2013

In Memory of Tom McIlwain (1940-2012)

IN MEMORY OF

Tom McIlwain

1940 - 2012

Tom McIlwain served as the Editor–in–Chief of the *Gulf Research Reports* (currently *Gulf and Caribbean Research*) from 1989 – 1994 when he served as Director of the Gulf Coast Research Laboratory (GCRL).

To read more about Tom and his life and service to the Lab, please visit our website at:
http://www.usm.edu/gcrl/about_us/Tom.McIlwain.php

Sand Bottom Microalgal Production and Benthic Nutrient Fluxes on the Northeastern Gulf of Mexico Nearshore Shelf

Jeffrey G. Allison
University of West Florida

M.E. Wagner
University of West Florida

M. McAllister
University of West Florida

A.K.J. Ren
University of West Florida

See next page for additional authors

Sand Bottom Microalgal Production and Benthic Nutrient Fluxes on the Northeastern Gulf of Mexico Nearshore Shelf

Authors

Jeffrey G. Allison, *University of West Florida*; M.E. Wagner, *University of West Florida*; M. McAllister, *University of West Florida*; A.K.J. Ren, *University of West Florida*; and R.A. Snyder, *University of West Florida*

SAND BOTTOM MICROALGAL PRODUCTION AND BENTHIC NUTRIENT FLUXES ON THE NORTHEASTERN GULF OF MEXICO NEARSHORE SHELF

Jeffrey G. Allison*, M. E. Wagner, M. McAllister, A. K. J. Ren, and R. A. Snyder
*University of West Florida, CEDB, 11000 University Parkway, Pensacola, FL 32514 USA; *Corresponding author, email: allisonjeffrey@hotmail.com*

ABSTRACT: Benthic microalgal production on the continental shelves may be an important contributor to the overall productivity of offshore ecosystems. We used light and dark benthic chambers to measure *in situ* production, respiration, and benthic nutrient flux on the nearshore quartzite sands of the northeast Gulf of Mexico shelf. Net exchange of O_2, NH_4^+, $NO_3^- + NO_2^-$, PO_4^{-3}, and SiO_2 was measured in samples taken from chambers at depths of 15 to 16 m offshore of Pensacola, FL. Phytoplankton production and respiration in near–bottom water was determined in paired light/dark BOD bottles to correct chamber measurements for water column processes. Sediment chlorophyll *a* (Chl *a*) averaged 4.8 μg Chl *a*/g. Phytoplankton averaged 5.5 μg Chl *a*/L. Pheophytin:chlorophyll ratios for the sediment were near 1 indicating an actively growing algal community. Phytoplankton net production ranged from 0.6 to 2.8 mg C/m^3/hr. Benthic net production in three separate determinations was 17.7 ± 6.1, 9.5 ± 2.9, and 8.8 ± 1.6 mg C/m^2/hr. Benthic respiration was 24.8 ± 0.7, 30.8 ± 1.4, and 11.3 ± 0.3 mg C/m^2/hr, respectively. Benthic gross production was thus 42.5 ± 5.2, 40.3 ± 1.2, and 20.3 ± 1.7 mg C/m^2/hr, respectively. Benthic nutrient fluxes were highly variable and generally low. Sediment uptake was observed for NH_4^+ and PO_4^{-3} throughout the study. $NO_3^- + NO_2^-$ and SiO_2 uptake was observed in 2004 with sediment release seen in 2005.

KEY WORDS: NPP, GPP, benthic chambers

INTRODUCTION

Benthic microalgae or the microphytobenthos are important primary producers in a wide variety of aquatic habitats including the nearshore continental shelf (Colijn and de Jonge 1984, Cahoon and Cooke 1992, Grippo et al. 2009, Jahnke et al. 2000). Cahoon and Cooke (1992) suggested that light and nutrient availability was sufficient to support benthic primary production beyond the North Carolina shelf break at 55 m depth, and Gattuso et al. 2006 showed that light availability is abundant in the coastal zone. Dynamics of offshore benthic community and phytoplankton production, respiration, algal biomass, and benthic nutrient fluxes have been examined in the northwestern Gulf of Mexico (GOM) impacted by the Mississippi River plume (Grippo et al. 2009, Baustian et al. 2011, Murrell and Lehrter 2011, Lehrter et al. 2012) but no reports are available for the sand bottom shelf of the Florida Panhandle Bight in the northeast GOM, an area extending about 240 km from Cape San Blas to Perdido Key.

Microphytobenthic communities are typically dominated by Bacillariophyceae (diatoms) encompassing various temperature, light, and salinity regimes. The contribution of benthic and epiphytic microalgae to shallow water ecosystem productivity has been well documented (Cahoon and Cooke 1992, Kang et al. 2003, Murrell et al. 2009), and supported by stable isotope tracer studies showing the importance of this production to benthic feeders (Sullivan and Moncreiff 1990, Kang et al. 2003). Some estimates have indicated benthic microflora production can equal or exceed phytoplankton production in the water column (Schreiber and Pennock 1995, Jahnke et al. 2000). However, relatively few studies have attempted to measure deeper water benthic microalgal production *in situ*, owing to the logistical difficulties involved.

Core sampling has most often been employed to measure benthic microalgal production (Glud 2008), but this technique involves potential disruption of the sediment–water interface and isolation of sediment sample from advective exchanges through pore waters (Huettel and Rusch 2000) that attenuate natural conditions. *In situ* incubation allows production and respiration to occur under natural light regimes, ambient temperature conditions with intact sediment structure, and permits pore water advection. Net DO fluxes can be derived by transforming the change of DO concentration in the benthic chamber into areal estimates for the sediment surface.

Difficulties with calculating biomass from cell abundance or counting microalgae has led researchers to quantify benthic microalgal biomass using chlorophyll *a* (Chl *a*) (MacIntyre et al. 1996, Grippo et al. 2009). Cahoon et al. (1990) found that up to 80% of the Chl *a* in Onslow Bay, NC was associated with the sediment, and benthic Chl *a* almost always surpassed the value of integrated water column Chl *a*. Similar results from sandy sediments on Ship Shoal, offshore Louisiana, were found by Grippo et al. (2009). Total pheopigments (Pheo *a*) to Chl *a* ratios have been used as an indicator of the physiological state of the microalgal community, with higher values indicating a stressed or declining community while lower numbers suggest an actively growing community (Lorenzen 1967, Grippo et al. 2009).

Benthic chambers have also been used to obtain estimates

of *in situ* benthic nutrient fluxes in shallow coastal systems and estuaries (Fisher et al. 1982, Nicholson et al. 1999). Microbial breakdown of organic material in sediments results in nutrient regeneration and an outward flux to the overlying water column. Sediments may also take up nutrients from the water column, but net flux is generally outward to the water column (Hopkinson et al. 2001, Lehrter et al. 2012). Nutrient regeneration at the benthic boundary layer from microbial processing of sedimented organic matter and mineralization of benthic microalgae by grazers are significant in buffering and recycling water column nutrients (Fisher et al. 1982) which supports planktonic production (Rowe and Phoel 1992). This study used diver–tended *in situ* benthic chambers to estimate microphytobenthic production, total benthic respiration, and nutrient flux rates on the nearshore continental shelf of the northeastern GOM.

MATERIALS AND METHODS

Study Area

The northeastern GOM shelf sediments along the Florida Panhandle Bight are dominated by coarse quartzite sands. The study site was located about 11 km south southeast of the Pensacola Bay, FL pass (30.25°N and 87.25°W) in 15–16 m water depth (Table 1). Diurnal tides (one high water and one low water occur during a tidal day) exist in the region with a microtidal range of <1.0 m (Oey 1995). Minimal bottom currents were reported by SCUBA divers during the 3 reported sampling events: 3 September 2004, 10 September 2004, and 25 July 2005.

Hydrographic Data

A handheld YSI® Model 85 was used to record profiles of water temperature, salinity, and dissolved oxygen (DO) during incubations. Incubation periods (Table 1) traversed the solar zenith, typically from 0900 to 1600 h. Irradiance measures ($\mu E/m^2/s$) were taken every hour during incubations with a Li–Cor® LI–190SA radiation sensor at the sea surface and at 1 m intervals in vertical profiles of the water column to the seafloor, allowing for the calculation of percent transmission of photosynthetic active radiation (%PAR) and light extinction coefficients (k_d) as: k_d = ln (surface irradiance/irradiance at depth)/depth.

Ambient water samples near the bottom were collected using a Van Dorn bottle. Sub–samples for dissolved nutrients (NH_4^+, NO_3^-, NO_2^-, PO_4^{-3}, SiO_2) were filtered through a 25 mm ashed (500°C for 1 h) Whatman® GF/F filter. Samples for pigment analysis were collected on 25 mm ashed GF/F filters, placed in foil envelopes, and analyzed by EPA Method 445.0 (US EPA 1992). All samples were kept on ice in the field and stored frozen (–80°C for pigments, –20°C for nutrients) until analysis.

Benthic Chambers

Replicate light and dark benthic chambers, 3 each, were constructed from clear acrylic domes of 0.26 m radius, 0.19

m height, covering an area of 0.212 m^2 with a volume of 0.027 m^3 (Figure 1A). Dark chambers were covered with 0.15 mm black polyethylene sheeting (Film–Gard®) to block light (Figure 1B). Stirring devices were assembled with hemispherical plastic cups transferring ambient external current to internal acrylic stirring paddles (Figure 1A). Domes were fitted with two barbed hose fittings and silicone tubing, one to allow sample water to be removed by syringe and the other to allow replacement water to enter chambers. Replacement water dilution of chamber water was assumed to be negligible. Four samples were taken from each replicate dome at each time
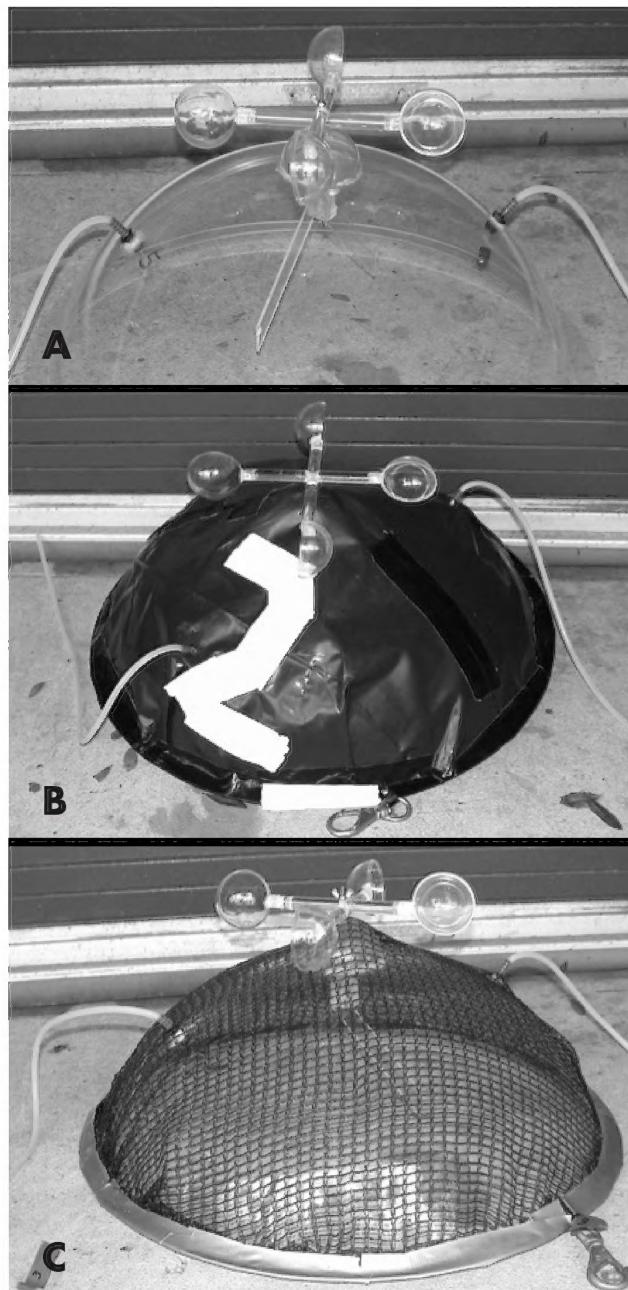

Figure 1. Light (**A**) and dark (**B**) benthic chambers with attached tubing for syringe sampling and current simulation device. Internal stirring paddles are visible in image (**A**). (**C**) Benthic chamber covered with shade cloth (30%) for the microphytobenthic response to variable irradiance compensation point estimation.

point (T_o and T_x = post—incubation) for DO and dissolved nutrients (DN). On deck, samples in syringes were processed for nutrient analysis as described above or for Winkler titrations as described below.

Chambers were carefully lowered and positioned by divers on the bottom to avoid sediment resuspension; chamber placement avoided macrophytes, large polychaete burrows, tunicates, and hard substrates. The chambers were secured to the bottom by piling sand over the dome's 2 in. external flange. We assumed that negligible water exchange occurred between inside and outside of the domes during incubation. Chambers were allowed to stabilize for 10—15 min before initial (T_o) samples were taken. On deck, DO samples were fixed with Winkler reagents by inserting blunt ended needles into the opening of the sampling syringes. Samples were mixed by inverting, capped, and stored on ice for transport to the laboratory.

Chamber Production and Respiration

Benthic and planktonic primary production and respiration were determined as changes in DO concentration by Winkler titration of 50 mL syringe samples or 300 mL BOD bottles, respectively (Eaton et al. 2005). To obtain benthic community production and respiration, water contained under domes was corrected for near bottom plankton production and respiration. Replicate (3) light and dark BOD bottles were filled on deck with bottom water collected in Van Dorn bottles using tubing placed into the bottom of each BOD bottle; water was allowed to overflow before sealing. BOD bottles were returned to the bottom and incubated in a rack placed near the benthic chambers. Three additional samples were fixed shipboard with Winkler reagents at the start of the incubation period. At the end of the incubation period, syringe samples were taken from the domes, and BOD bottles were collected, and fixed on deck within 15 min of sampling. Winkler titrations were completed at the lab within 12 h of sample collection.

Net benthic primary production (NBPP, mg C/m^2/h) was calculated from DO concentration changes in light and dark chambers using the formula for non—standard sample volumes (Strickland and Parsons 1972):

NBPP = ((([DO]t_x – [DO]t_o) * V * 12) / (PQ * H * A),

where: [DO]t_x = dissolved oxygen concentration in mmol/L at the end of incubation; [DO]t_o = dissolved oxygen concentration in mmol/L at the beginning of incubation; V = volume of benthic chamber in liters; 12 = atomic weight of carbon; PQ = photosynthetic quotient (mol O_2 evolved / mol C fixed), 1.2 is recommended (Strickland and Parsons, 1972); H = incubation time in h; and A = area under the benthic chamber in m^2.

Benthic respiration was also determined using this equation with the exception that a respiration quotient (RQ) of 1.0 was used in place of the PQ, as recommended by Strickland and Parson (1972).

Student's t—test was performed to determine if initial DO concentrations were significantly different from post incubation measures. Confidence intervals (95%) were determined for benthic flux data in order to establish if a flux was significantly different from zero.

Algal Biomass

Chlorophyll analyses were conducted on sediment samples (collection techniques described in Allison 2006) and on filtered water samples to represent microphytobenthic and phytoplankton biomass, respectively. Samples were extracted in 8—10 mL of 90% acetone with probe sonication and held overnight at −20°C. Extracts were measured for their fluorescence on a Perkin Elmer® LS45 luminescence spectrometer according to the methods of Dandonneau and Neveux (2002), beginning at excitation wavelengths of 406 nm. Emission (fluorescence) wavelengths for detection of Chl a were between 666 and 668 nm, for Chl c at 630 nm, and for Pheo a from 646 to 656 nm. Pheophytin a was quantified by acidifying the stock Chl a (Sigma®) standard according to the methods of Welschmeyer (1994). Chl c was quantified using standard material extracted from reference Cryptophyceae (DHI Water and Environment®). Pigment concentrations in acetone extracts were normalized to dry weight for sediments and to volume for water samples.

Nutrient Analyses

All samples for nutrient analysis (SiO_2, PO_4^{-3}, NH_4^+, NO_3^- + NO_2^-) were analyzed using standard methods (US EPA 1993) on a Bran—Luebbe® AutoAnalyzer 3. Benthic nutrient fluxes were calculated from nutrient results as in Rowe and Phoel (1992): Flux = [(Initial Conc. – Final Conc.) * Vol.] / (Time * Surface Area), expressed in mmol/m^2/d.

Irradiance Curve for Benthic Production

A separate experiment was conducted on 25 July 2005 to measure the microphytobenthic response to variable irradiance allowing compensation point estimations (P/R = 0). Benthic chambers were covered with a commercial (Greenhouse Mega Store®) shade cloth with 30% and 60% shading density (n = 3 each; Figure 1C). Irradiance intensity for each treatment was recorded by a Li—Cor® light meter and plotted against the measured benthic production. Samples were collected and processed for DO flux calculations as described above.

Results

Bottom water salinity was similar on all sampling dates, ranging from 33—36 (Table 1). Bottom water temperature ranged from 26—30°C (Table 1). DO on the bottom was lowest (3.23 mg/ L) on 25 July 2005 (Table 1). YSI® temperature and salinity profiles indicated that the water column was well—mixed on 3 September 2004, and stratified on the other 2 dates (data not shown). Percent surface PAR at the bottom averaged 7.5 ± 0.01% (mean ± sd) with a range of 1.9 –10.5%. Based on surface irradiance values, mean light

TABLE 1. *Summary of sampling dates, incubation times, daylight hours, depth, physicochemical properties and the light extinction coefficient (kd) from PAR attenuation with depth from each experiment.*

Date	Incubation Duration (h)	Total Daylight (h)[1]	DO (mg/L)	Salinity	Temp (°C)	Light Extinction (kd/ m)	Depth (m)
3 Sep 04	4	12.7	8.95	33.1	30.3	0.17	15
10 Sep 04	3.5	12.5	5.08	35.6	26.6	0.16	16
25 Jul 05	5	13.8	3.23	34.2	26.9	0.19	15

[1]taken from: http://aa.usno.navy.mil/data/docs/Dur_OneYear.php/

intensities reaching the bottom for the three incubations were 7.3, 4.8, and 8.0 µE/m^2/s, chronologically. The mean k$_d$ value for the site was 0.17 ± 0.01/m and ranged from 0.16 – 0.19/m for all experiments (Table 1). Divers noted good visibility on the bottom for both 2004 experiments and reduced visibility, presence of planktonic jellyfish, and a more disturbed bottom topography for the 2005 experiment.

Bottom water nutrient concentrations varied 6–fold among sample dates. The highest concentrations of dissolved inorganic nitrogen (DIN = NO$_3^-$ + NO$_2^-$ + NH$_4^+$) occurred in 2005. DIN was mostly NH$_4^+$ ranging from 51%–84% of DIN. For dissolved inorganic phosphorous (DIP = PO$_4^{-3}$), concentrations varied from 0.06–0.25 µM, being greatest on 10 September 2004 (Table 2). Bottom water SiO$_2$ concentrations were similar on all sample dates, ranging from 4.07–5.36 µM (Table 2).

The largest DIN fluxes were measured in 2004 experiments (Table 3). Sediment DIN uptake was observed in all experiments except in light domes on 3 September 2004. On the other dates, DIN uptake in light domes was twice as high as dark domes. DIN flux was dominated by NH$_4^+$ for all experimental treatments. Release of NO$_3$ + NO$_2$ from sediments occurred in both light and dark domes in 2005, with

measures twice as high as light domes. Sediment release of SiO$_2$ happened on 25 July 2005, also with dark incubations twice as high as light domes (Table 3).

Plankton net production was greatest (2.8 mg C/m^3/h) on 3 September 2004 and nearly identical on 25 July 2005 (Table 4). On 10 September 2004, net plankton production (0.74 ± 0.2 mg C/m^3/h) and respiration (1.01 ± 0.1 mg C/m^3/h) were much lower resulting in the lowest recorded GPP (Table 4). Plankton respiration was greatest (3.58 ± 0.4 mg C/m^3/h) on 3 September 2004 which was 3.5 times more than the experiment the following week (Table 4). Phytoplankton biomass, as chlorophyll, was less abundant during 2004 incubations than in 2005 (Table 5). Pheo a always exceeded Chl a in both sediment and phytoplankton samples (Table 5). The Chl a : Pheo a ratio was similar for both phytoplankton and sediments with an overall range of 0.60–0.77. Sediment Chl c estimates were between 12–15% of Chl a values while phytoplankton Chl c estimates were slightly higher, between 15–25%, indicating only a small presence of the Bacillariophyceae.

Net benthic production was greatest on 3 September 2004 (17.7 ± 6.1 mg C/m^2/h). The value obtained 10 September 2004 was nearly half, and was similar to the value obtained the following year (25 July 2005; 8.8 ± 1.6 mg C/m^2/h) (Table 4). Net benthic production for all experiments in the study period ranged from 6.6–23.8 mg C/m^2/h. Benthic respiration ranged from 10.8–38.6 mg C/m^2/h, was the largest on 10 September 2004 (30.8 ± 1.4 mg C/m^2/h), slightly less on 3 September 2004 (24.8 ± 0.7 mg C/m^2/h), and less than half the following year on 25 July 2005 (11.3 ± 0.3 mg C/m^2/h). GPP always exceeded benthic respiration and was almost equal on the 2004 events (42.5 ± 5.2 and 40.3 ± 1.2 mg C/m^2/h for experiments 1 and 2, respectively) which were double that of 25 July 2005 (20.3 ± 1.7 mg C/m^2/h) (Table 4).

Benthic production response to irradiance was measured on 25 July 2005 and showed a linear (r = 0.99) relation-

TABLE 2. *Ambient bottom water dissolved nutrient concentrations for each experiment. All concentrations are expressed as µM. DIN = dissolved inorganic nitrogen; DIP = dissolved inorganic phosphorous; SiO$_2$ = dissolved silica.*

Date	DIN	NO$_3^-$+NO$_2^-$	NH$_4^+$	DIP	SiO$_2$
3 Sep 04	3.60	0.59	3.01	0.16	5.16
10 Sep 04	2.16	0.59	1.58	0.25	4.07
25 Jul 05	12.37	6.13	6.24	0.06	5.36

dark domes releasing twice as much as light domes. However, the largest sediment release was in light chambers on 3 September 2004 (Table 3). PO$_4^{-3}$ was released from sediments on 3 September 2004 and sediment uptake observed on the other 2 dates with −7.5 µmol/m^2/h on 10 September 2004 as the largest PO$_4^{-3}$ flux measured. Sediment uptake of SiO$_2$ occurred on 10 September 2004 with dark dome

ship for the replicate clear dome and domes covered with shade cloth (Figure 2). The bottom irradiance of 134 µE/m^2/s represented 8% transmission of surface irradiance. Irradiance under the shade cloths was 42.3 and 24.8 µE/m^2/s, respectively for the 2 treatments (30 and 60% shade) representing 2.5 and 1.5% transmission of surface irradiance. The compensation irradiance, or point of zero net community

TABLE 3. *Average benthic flux of dissolved nutrients (µmol/m²/h) from light and dark chambers (n=3 each) measured in situ for each experiment. Negative values denote uptake by sediments. * = significantly different from zero; DIN = dissolved inorganic nitrogen; $NO_3^- + NO_2^-$ = nitrate + nitrite; NH_4^+ = ammonia; PO_4^{-3} = dissolved inorganic phosphorous; SiO_2 = dissolved silica.*

Date	Dome	DIN	$NO_3^- + NO_2^-$	NH_4^+	PO_4^{-3}	SiO_2
3 Sep 04	Light	197.6 ± 24.6*	21.2 ± 5.1*	176.4 ± 19.5*	4.1 ± 3.1	49.1 ± 23.4*
3 Sep 04	Dark	-277.1 ± 299.2	-0.3 ± 0.6	-276.8 ± 299.4	3.6 ± 2.3	-144.6 ± 99.5
10 Sep 04	Light	-322.3 ± 94.7*	-9.5 ± 4.4*	-312.8 ± 95.6*	-5.5 ± 0.4*	-86.6 ± 24.3*
10 Sep 04	Dark	-118.8 ± 74.2	-6.5 ± 0.8*	-114.0 ± 76.5	-7.5 ±1.4*	-154.3 ± 6.9*
25 Jul 05	Light	-86.7 ± 22.3	8.6 ± 2.4	-95.3 ± 33.8*	-0.7 ± 0.2*	113.4 ± 37.2*
25 Jul 05	Dark	-40.2 ± 11.4*	15.6 ± 2.7	-55.8 ± 17.2*	-0.1 ± 0.1	226.8 ± 24.6*

TABLE 4. *Summary data on the net production of the benthos and plankton (NPP) measured in light domes and BOD bottles, respectively. Benthic and planktonic respiration (RESP) measured in dark domes and BOD bottles, respectively. Gross production (GPP) is the total of NPP and RESP for all experiments. Hourly means are mean ± sd. P/R values are instantaneous measures and are not indicative of overall net community P/R for the shelf.*

|	BENTHOS mg C/m²/h				PLANKTON mg C/m³/h		
Date	NPP	RESP	GPP	P/R ratio	NPP	RESP	GPP
3 Sep 04	11.6	25.7	37.3	1.5	2.8	3.8	6.6
3 Sep 04	23.8	23.9	47.7	2.0	--	3.3	6.1
Mean	17.7 ± 6.1	24.8 ± 0.7	42.5 ± 5.2	--	--	3.5 ± 0.4	6.4 ± 0.4
10 Sep 04	12.5	29	41.5	1.4	0.6	0.9	1.7
10 Sep 04	6.6	32.6	39.2	1.2	0.9	1.1	2.0
Mean	9.5 ± 2.9	30.8 ± 1.4	40.3 ± 1.2	--	0.7 ± 0.2	1.0 ± 0.1	1.7 ± 0.3
25 Jul 05	10.4	11.7	22.1	1.9	2.8	1.2	4.0
25 Jul 05	7.2	11.5	18.7	1.6	--	2.6	5.4
25 Jul 05	--	10.8	--	--	--	2.2	5.0
Mean	8.8 ± 1.6	11.3 ± 0.3	20.3 ± 1.7	--	--	2.0 ± 0.7	4.8 ± 0.7
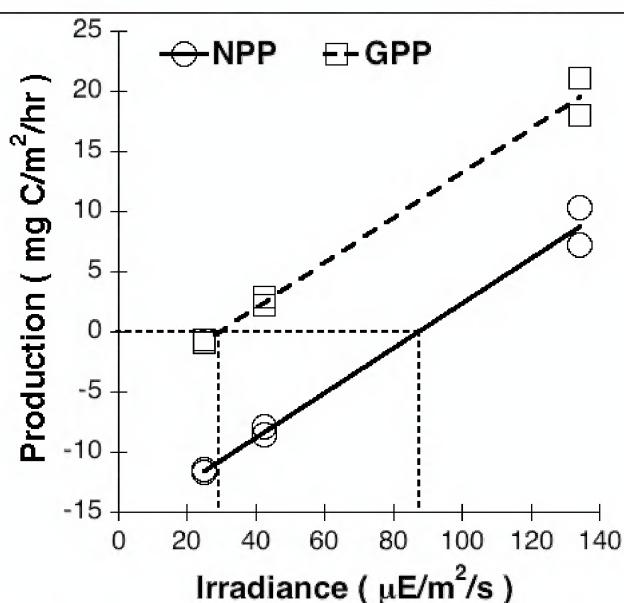
Figure 2. *Benthic net (NPP) and gross (GPP) community production response to irradiance intensity during a single experiment on 25 July 2005. The least squares regression model slope is 0.186 ± 0.01, r^2 = 0.989. Dotted lines indicate the point of zero net and gross community production, respectively.*

production, estimated from regression analysis of benthic chamber production versus irradiance was 87 µE/m²/s, or 5.2 % of surface irradiance (Figure 2). GPP was zero at 29 mE/m²/s.

DISCUSSION

The majority of continental shelves in the world are shallow enough that the benthos is included in the photic zone (Gattuso et al. 2006). With distance away from coastal enrichment sources, water column turbidity decreases which increases light penetration. Increasing depth with distance may be offset by lower k_d values with distance offshore so that the potential for benthic primary production exists for much of the continental shelf ecosystem. This was certainly the case for this study of the nearshore shelf of the Florida Panhandle Bight, providing the first estimates of benthic community production, respiration, and benthic nutrient fluxes on the nearshore continental shelf of the northeastern GOM.

One major advantage of the benthic chamber method is that the area of sediment enclosed by chambers can be large, minimizing the effect of micro–heterogeneity. Theoretically, *in situ* benthic chambers should give the most accurate estimate of fluxes, because chambers include the effects of diffusion and incorporate the exchange across the sediment–water interface created by bioturbation. Diver deployment of simple benthic chambers can be a major disadvantage when considering weather, water depth, water temperature, wind speed, current velocity, water visibility, bottom time limitations, and risk to personnel. In order for benthic chamber experiments to properly estimate benthic primary production, one must assume that all samples are taken from homogeneous, well–mixed chamber water (Glud 2008, Lehrter et al. 2012). The stirring rate inside the chamber should mimic the ambient near bottom water flow to avoid build up of concentration gradients that would inhibit diffusive fluxes and cause uneven sampling.

In our observations, elevated bottom water N and low P

TABLE 5. *Summary of the mean (± sd) and range of photopigment values for sediment (µg/g DW) and phytoplankton (µg/L). Pheophytin a is the main chlorophyll a degradation product.*

	Chl *a*	Chl *c*	Pheo *a*	Chl *a* : Pheo *a*
Sediments				
10 Sep 04	4.77 ± 0.84	0.63 ± 0.12	6.81 ± 1.26	0.69 ± 0.01
Range	2.46 – 8.71	0.31 – 1.17	3.41 – 12.79	0.64 – 0.77
Phytoplankton				
25 Jul 05	5.48 ± 1.24	1.37 ± 0.40	8.30 ± 2.38	0.68 ± 0.04
Range	3.45 – 7.74	0.82 – 2.14	4.87 – 12.87	0.60 – 0.71
03 Sep 04	0.71 ± 0.06	0.10 ± 0.02	0.85 ± 0.14	0.83 ± 0.12
Range	0.64 – 0.77	0.07 – 0.14	0.67 – 1.12	0.70 – 0.91

concentrations in 2005 compared to 2004 (see Table 2), may suggest that production was P–limited. The range of PO_4^{-3} flux in this study was much less in both directions (−8 to +4 µmol/m²/h) than the range (−0.03 to +0.50 mmol/m²/h) reported by Hopkinson et al. (2001) in offshore waters of Massachusetts Bay, but more similar to the range (−1.5 to +9.6 µmol/m²/h) reported in Reay et al. (1995) from the nearshore Delmarva Peninsula. Sediment release (in 2005 only) of SiO_2 was less than the range reported (1.8–14.1 mmol/m²/h) for the offshore waters of Massachusetts Bay (Hopkinson et al. 2001). Hopkinson et al. (2001) did not report any findings of sediment uptake for SiO_2 as was observed in the current investigation on 10 September 2004. Jahnke et al. (2000) reported a SiO_2 release from southeastern U.S. continental shelf sediments, but concentration changes in chamber water of NH_4^+, NO_3^- + NO_2^-, and PO_4^{-3} were not significant. The highest SiO_2 release in the current study (227 µmol/m²/h) was comparable to the high value (219 µmol/m²/h) seen by Marinelli et al. (1998) for the South Atlantic Bight.

Marinelli et al. (1998) and Murrell et al. (2009) in Pensacola Bay found that nutrient fluxes in light versus dark chambers were highly variable and did not follow a regular shallow water pattern of increased efflux in the dark and decreased efflux in the light, similar to our experience. This pattern was true for SiO_2 regardless of flux direction. The opposite was found for NH_4^+ and DIN fluxes with increased concentrations in light chambers. The range of NH_4^+ flux in this study was greater in both directions than the range reported (−13.9 to +59.5 µmol/m²/h) for the nearshore Georgia shelf (Marinelli et al. 1998). NH_4^+ was predominantly taken up by sediments, which was contrary to the findings of Reay et al. (1995).

Reduced respiration rates and GPP on 25 July 2005 may

have been the result of reduced visibility and disturbed bottom as reported by divers and not by % PAR reduction with cloud cover, as the k_d value for that sampling event was similar to those from the previous year. Although GPP was reduced by half in that incubation, net benthic production was nearly identical to 10 September 2004. Benthic production was also 2 times greater on the first experiment with less respiration than the second experiment one week later. Increased respiration on 10 September 2004 made up the difference when comparing the similar GPP estimates of the 2004 experiments. Reduced ambient oxygen concentrations on the bottom on 25 July 2005 may explain the reduced production observed on that date.

The range of net benthic production, respiration, and GPP in this study was comparable to values of the nearshore study sites in Onslow Bay, NC (Cahoon and Cooke 1992). Average hourly respiration rates obtained by Cahoon and Cooke (1992), 18.2 mg C/m²/h, are comparable to the 2004 data and slightly higher than 2005. Estimates in this study were much greater than values of benthic production (1.8–8.5 mg C/m²/h) and respiration (0.6–6.5 mg C/m²/h) reported for tropical marine sediments (Bunt et al. 1972).

Patchy distribution of benthic microalgae can lead to a 2–to 10–fold difference in biomass estimates over a distance of a few centimeters (Colijn and de Jonge 1984). Thus, a wide range can be expected when estimating sediment microalgal processes. Sediment Chl *a* values in the current investigation were comparable to the shallow sands of the Swan–Canning estuary, Australia which ranged between 2–20 µg Chl *a*/g (Masini and McComb 2001), and slightly higher than the LCS studies of Lehrter et al. (2012) and Baustian et al. (2011) reporting maximum values of 2 µg/g.

Photopigments (Chl *a*, Chl *c*) and the primary degradation product (Pheo *a*) followed similar trends and showed similar variation in all experiments. Chl *a* : Pheo *a* ratios were similar for the water column and benthos, and values near 1 for all samples indicate an actively growing assemblage. Benthic diatoms were visually observed in sediment grab samples from the continental shelf further offshore (30 km) of Pensacola, FL at a depth of 67 m (Allison, unpublished data), indicating the potential for benthic autotrophic production to occur over broad areas on the northeastern GOM continental shelf.

Literature Cited

Allison, J.G. 2006. Dynamics of estuarine microphytobenthos in a shallow water sand bottom habitat. M.S. thesis. University of West Florida, Pensacola, FL, USA, 110 p.

Bunt, J.S., C.C. Lee, and E. Lee. 1972. Primary productivity and related data from tropical and subtropical marine sediments. Marine Biology 16:28–36.

Cahoon, L.B. and J.E. Cooke. 1992. Benthic microalgal production in Onslow Bay, North Carolina, USA. Marine Ecology Progress Series 84:185–196.

Cahoon, L. B., R. S. Redman, and R. S. Tronzo. 1990. Benthic microalgal biomass in sediments of Onslow Bay, North Carolina. Estuarine and Coastal Shelf Science 31: 805–816.

Colijn, F. and V. de Jonge. 1984. Primary production of microphytobenthos in the Ems–Dollard estuary. Marine Ecology Progress Series 14:185–196.

Dandonneau, Y. and J. Neveux. 2002. An assessment of biases in flourometric estimates of chlorophyll and pheophytin concentration in seawater, by reference to fluorescence spectral characteristics of main chlorophyllous pigments and large scale sampling. Instrumental Notes of IPSL 18:1–19.

Eaton, A.D., L.S. Clesceri, E.W. Rice, and A.E. Greenberg (Eds.). 2005. *Standard Methods for the Examination of Water and Wastewater* (18th ed.). American Public Health Association, Washington, DC, USA, 1100 p.

Fisher, T.R., P.R. Carlson, and R.T. Barber. 1982. Sediment nutrient regeneration in three North Carolina estuaries. Estuarine and Coastal Shelf Science 14:101–116.

Gattuso, J.–P., B. Gentili, C.M. Duarte, J.A. Kleypas, J.J. Middelburg, and D. Antoine. 2006. Light availability in the coastal ocean: impact on the distribution of benthic photosynthetic organisms and their contribution to primary production. Biogeosciences 3:489–513.

Glud, R.N. 2008. Oxygen dynamics of marine sediments. Marine Biology Research 4:243–289.

Grippo, M.A., J.W. Flegger, R.Condrey, and K.R. Carman. 2009. High benthic microalgal biomass found on Ship Shoal, north–central Gulf of Mexico. Bulletin of Marine Science 84:237–256.

Hopkinson, C. S., A. E. Giblin, and J. Tucker. 2001. Benthic metabolism and nutrient regeneration on the continental shelf of Eastern Massachusetts, USA. Marine Ecology Progress Series 224:1–19.

Huettel, M. and A. Rusch. 2000. Transport and degradation of phytoplankton in permeable sediment. Limnology and Oceanography 45:534–549.

Jahnke, R.A., J.R. Nelson, R.L. Marinelli, and J E. Eckman. 2000. Benthic flux of biogenic elements on the Southeastern US› continental shelf: Influence of pore water advective transport and benthic microalgae. Continental Shelf Research 20:109–127.

Kang, C.K., J.B. Kim, K.S. Lee, P.Y. Lee, and J.S. Hong. 2003. Trophic importance of benthic microalgae to macrozoobenthos in coastal bay systems in Korea: dual stable C and N isotope analyses. Marine Ecology Progress Series 259: 9–92.

Lehrter, J.C., D.B. Beddick, R. Devereux, D.F. Yates, and M.C. Murrell. 2012. Sediment–water fluxes of dissolved inorganic carbon, O_2, nutrients, and N_2 from the hypoxic region of the Louisiana continental shelf. Biogeochemistry 109:233–252.

Lorenzen, C.J. 1967. Vertical distribution of chlorophyll and phaeo–pigments: Baja California. Deep–Sea Research 14:735–745.

MacIntyre, H.L., R.J. Geider, and D.C. Miller. 1996. Microphytobenthos: The ecological role of the "Secret Garden" of unvegetated, shallow–water marine habitats. I. Distribution, abundance and primary production. Estuaries 19:186–201.

Marinelli, R.L., R.A. Jahnke, D.B. Craven, J.R. Nelson, and J.E. Eckman. 1998. Sediment nutrient dynamics on the South Atlantic Bight continental shelf. Limnology and Oceanography 48:1305–1320.

Masini, R.J. and A.J. McComb. 2001. Production by microphytobenthos in the Swan–Canning estuary. Hydrobiological Processes 15:2519–2535.

Murrell, M.C., J.G. Campbell, J.D. Hagy and J.M. Caffrey. 2009. Effects of irradiance on benthic and water column processes in a Gulf of Mexico estuary: Pensacola Bay, Florida, USA. Estuarine, Coastal, and Shelf Science 81:501–512.

Murrell, M.C. and J.C. Lehrter. 2011. Sediment and lower water column oxygen consumption in the seasonally hypoxic region of the Louisiana continental shelf. Estuaries and Coasts 34:912–924.

Nicholson, G.J., A.R. Longmore, and W.M. Berelson. 1999. Nutrient fluxes measured by two types of benthic chamber. Marine and Freshwater Research 50:567–572.

Oey L. 1995. Eddy and wind–forced shelf circulation. Journal of Geophysical Research 100:8621–8637.

Reay, W.G., D.L. Gallagher, and G.M. Simmons. 1995. Sediment–water column oxygen and nutrient fluxes in nearshore environments of the lower Delmarva Peninsula, USA. Marine Ecology Progress Series 118:215–227.

Rowe, G.T. and W.C. Phoel. 1992. Nutrient regeneration and oxygen demand in Bering Sea continental shelf sediments. Continental Shelf Research 12:439–449.

Schreiber, R.A. and J.R. Pennock. 1995. The relative contribution of benthic microalgae to total microalgal production in a shallow sub–tidal estuarine environment. Ophelia 42:335–352.

Strickland, J.D.H. and T.R. Parsons. 1972. A practical handbook of seawater analysis. Fisheries Research Board of Canada, Vol. 167. Ottawa, Ontario, Canada, 310 p.

Sullivan, M.J. and C.A. Moncreiff. 1990. Edaphic algae are an important component of salt marsh food webs: Evidence from multiple stable isotope analysis. Marine Ecology Progress Series 62:149–159.

U.S. Environmental Protection Agency. 1992. Methods for the determination of chemical substances in Marine and Estuarine Environmental Samples. U.S. Environmental Monitoring Laboratory, Office of Research and Development, EPA/600/R–92/121, Cincinnati, OH, 102 p.

U.S. Environmental Protection Agency. 1993. Methods of determination of inorganic substances in environmental samples. U.S. Environmental Monitoring Laboratory, Office of Research and Development EPA/600/R–93/100, Cincinnati, OH, 535 p.

Welschmeyer, N.A. 1994. Fluorometric analysis of chlorophyll a in the presence of chlorophyll b and pheopigments. Limnology and Oceanography 39:1985–1992.

What is Known About Species Richness and Distribution on the Outer-Shelf South Texas Banks?

Harriet L. Nash
Texas A&M University, Corpus Christi

Sharon J. Furiness
Texas A&M University, Corpus Christi

John W. Tunnell Jr.
Texas A&M University, Corpus Christi

WHAT IS KNOWN ABOUT SPECIES RICHNESS AND DISTRIBUTION ON THE OUTER—SHELF SOUTH TEXAS BANKS?

Harriet L. Nash*, Sharon J. Furiness, and John W. Tunnell, Jr.

*Harte Research Institute for Gulf of Mexico Studies, Texas A&M University–Corpus Christi, 6300 Ocean Drive, Unit 5869, Corpus Christi, TX 78412 USA; *Corresponding author, email: harriet.nash@tamucc.edu*

ABSTRACT: The outer—shelf South Texas Banks, also known as the snapper banks, are known by fishermen to be excellent fishing grounds. However, few scientific studies have been conducted to describe the ecology of these uncommon but distinctive habitats. This paper describes results of a literature review and an assessment to determine what is known about the biota of the South Texas Banks and to assist in developing renewed interest and focus on these topographic highs. The outer—shelf South Texas Banks include relict coralgal reefs and relict barrier islands, and we also include data for a nearshore site, which is geographically and geologically separated from the offshore banks. Obtainable scientific literature was reviewed, and biodiversity data were compiled. Results indicate that one of the most studied sites, Southern Bank, could be used as a surrogate to describe potential biodiversity at other, less studied South Texas Banks. Conclusions support the need for more biological studies at all of the South Texas Banks. Results of future studies, when combined with existing results, could be used to identify sites as potential candidates for place—based protection.

KEY WORDS: Gulf of Mexico, hard banks, drowned coralgal reef, relict barrier island

INTRODUCTION

The continental shelf of the northwestern Gulf of Mexico (GOM) is very broad and primarily covered with soft sand or clay sediment. Hard banks and reefs with conspicuous topographic relief occur uncommonly on the otherwise flat outer continental shelf. Such offshore sites have been known for almost 100 years (Carsey 1950, Geodicke 1955), but biological data are scarce for all but West and East Flower Garden Banks and Stetson Bank (Tunnell et al. 1978, Earle 1998, Tunnell et al. 2005). Offshore hard banks are geographically and geologically divided into two broad categories, separated by a line extending seaward from Matagorda Bay to the shelf edge (Figure 1, Rezak et al. 1990, Holcombe et al. 2010). Northeast of this line, bathymetric highs form as a result of salt diapirs on the shelf edge and upper continental slope. Salt diapirs are formed from sheets of allochthonous salt beneath the seafloor when faults or high differential pressure cause the fluid salt to move upward through the seafloor (Roberts 2011). Southwest of the line, allochthonous salt sheets are not present, and seismic signatures reveal dense stratigraphic sediments that indicate the presence of drowned barrier islands and coral reefs on the mid and outer shelf (Berryhill 1986). The drowned coralgal reefs off South Texas have more topographic relief than the relict barrier island ridges and, therefore, provide habitat for many species, which will be the main focus of this paper. Two recent opportunities, in 2006 and 2008, to visit several of these banks, map them with multi—beam imagery, and survey them with a submersible remotely operated vehicle (ROV) have renewed interest in these little—known but important topographic highs (Tunnell et al. 2009, Weaver et al. 2009).

Figure 1. Locations of selected hard banks and reefs of the northwestern Gulf of Mexico. The dotted line represents estimated geological separation of the types of banks on the continental shelf in the northwestern Gulf of Mexico. The sites southwest of the line are the South Texas Banks according to Rezak et al. (1985).
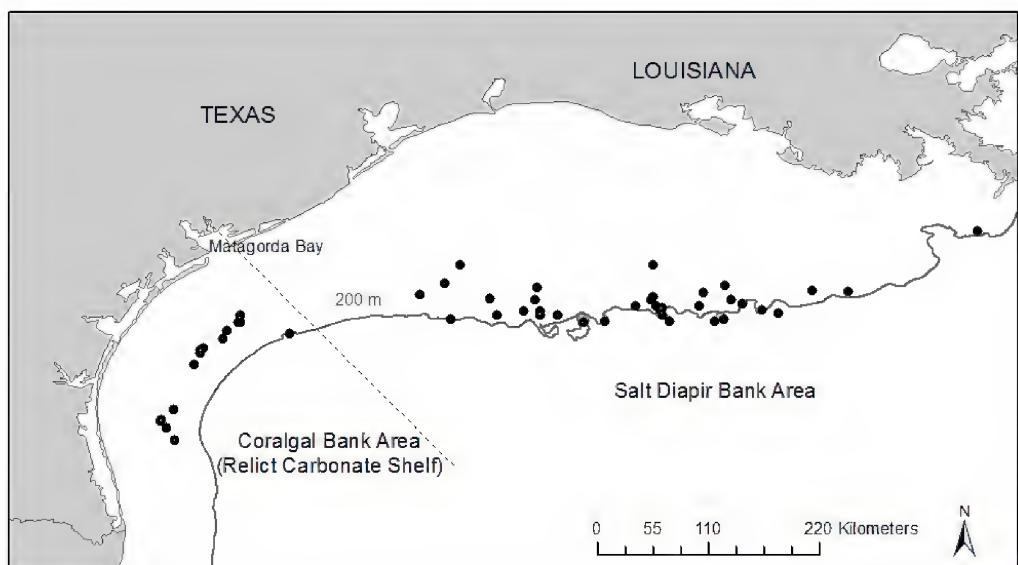
The broad, gently sloping continental shelf off South Texas is composed of terrigenous sediments. Across the South Texas shelf, a sea surface temperature gradient exists during winter months, with a low nearshore temperature of 14°C and a minimum outer shelf water temperature of 19–20°C (Flint and Rabalais 1981). In winter, the water temperature is fairly consistent throughout the water column, but thermal stratification is present in summer, particularly in deep waters. Salinity is typically high except when the Mississippi River plume covers the shelf during spring and early summer and lowers salinity in the top 20–30 m of the water column (Flint and Rabalais 1981). Freshwater inflow, from rainfall as well as Texas rivers and the Mississippi River, is a major factor that influences salinity, temperature, and ecological communities in the shallow waters of the inner shelf. Mid– to outer–shelf waters contain over 20 rare hard–substrate features of topographical relief (Berryhill 1986, Belopolsky and Droxler 1999). Many of these hard–bottom habitats are relict carbonate reefs. For purposes of this paper, we collectively refer to the relict reefs and relict barrier islands as the outer–shelf South Texas Banks.

Many technical government reports were published in the 1970s in support of petroleum exploration in the GOM, but since then, few studies have targeted the South Texas Banks. These banks, also referred to as the snapper banks by local fishermen, comprise several topographically distinct features on the continental shelf in the GOM off the coast of Texas south of Matagorda Bay. The vertical relief of the banks provides habitat complexity that attracts a variety of marine fauna and sustains communities that are not typical throughout the majority of the continental shelf, which is mostly bathymetrically planar with sandy or muddy substrate. The purpose of this study is to assess current knowledge and data gaps regarding the biota of the South Texas Banks. The goal is to use the results to guide future studies of the biodiversity and ecological interactions of the cluster of outer–shelf banks that provide rare hard–substrate habitat on the South Texas continental shelf.

METHODS

The authors comprehensively reviewed all information that could be obtained related to biota known to oc-

cur at the South Texas Banks. Information sources included peer–reviewed publications, government and non–government technical reports, master's theses, doctoral dissertations, unpublished papers, and unpublished data. Next, a taxonomic database was created to link organism occurrences to specific sites or, in some cases, to the collective group of South Texas Banks. Most taxa were identified to the level of genus, but several were identified only to family or, occasionally, only to order. For a more accurate representation of taxonomic diversity at the South Texas Banks, analysts sorted and filtered presence/absence data to determine the number of taxa that could be identified to the level of species. Site–specific taxonomic data, including those identified by this project, are in the process of being added to the Biodiversity of the Gulf of Mexico Database, an online resource available at GulfBase.org (Moretzsohn et al. 2013). Finally, the authors performed a case study of Southern Bank (27°26.5'N, 96°31.5'W; Figure 2) because it is one of the most studied sites of the outer–shelf South Texas Banks and exemplifies the kind and abundance of biota that might be present at


Figure 2. Locations of selected South Texas Banks.

less studied sites in the area depending on their biophysical characteristics.

Study Area Description

The South Texas Banks that are closest to the shelf edge consist of hard, carbonate substrate topped with fine sediment and have geologic signatures indicating that they are relict Pleistocene coralgal reefs (Rezak et al. 1985, Berryhill 1986). Such findings are curious because conditions in the GOM were less favorable for corals during the Pleistocene, when the last deglaciation occurred, than they are today. Compared to current conditions in the GOM, Pleistocene sea surface temperatures and salinity were lower, and sea–level rise was much faster (Belopolsky and Droxler 1999). Corals experience stress or even mortality when water temperature and salinity drop below their tolerance ranges or when water depth increases enough to prevent ample light from reaching the coral's symbiotic, photosynthesizing zooxanthellae. Regardless, geological data from the 1970s and 1980s indicate that coralgal reefs flourished on the edge of the continental shelf off South Texas. Holcombe et al. (2010) state that corals settled on bathymetric highs and formed reefs that flourished during the early Holocene when conditions were appropriate (higher water temperatures, lower sea level, and terrigenous sediments that did not flow directly to locations of the South Texas Banks). Conditions changed during the late Holocene resulting in reef die–offs. Seismic profiles indicate that about the lower two thirds of topographic relief of the banks were buried in clay and mud after the drowning of the reefs (Belopolsky and Droxler 1999). However, the shallower portions of the Holocene reefs remain as the South Texas Banks, which continue to provide rare habitat today on an otherwise flat continental shelf of soft substrates.

In addition to the sites about 60–70 km offshore near the shelf edge, several low–relief, flat–topped structures along the 20–30 m contour and about 20–30 km offshore have been interpreted as relict barrier island ridges (e.g., Seabree Banks, Steamer Bank; Berryhill 1976), but little is known of their biota and ecology. Closer to shore, Seven and One–Half Fathom Reef (7.5 Fathom Reef) is one of the most studied. It is located within state waters, while the other South Texas Banks are closer to the shelf edge in Federal waters (Figure 2). Seven and One–Half Fathom Reef is located about 3.2 km from shore almost directly west of Blackfish Ridge and differs geologically from the other South Texas Banks in that it is of lacustrine origin as evidenced by its Pleistocene terrestrial fossils (Tunnell and Causey 1969). Fossils include teeth and bone fragments from terrestrial mammals as well as freshwater and land snails. Analysis of rock samples from 7.5 Fathom Reef identified sediments of old coastal plain formations, thus confirming the lacustrine origin of the ridge (Thayer et al. 1974). Sample features, such as root tubules and soil glaebules, indicative of soil develop-

ment and calichification, are likely a result of fluctuations in the water table during the Pleistocene (Thayer et al. 1974). The calichified sediments became hardened and were not affected strongly by the Holocene transgression and remained as the topographic prominence now known as 7.5 Fathom Reef.

The number of South Texas Banks is debatable, depending on how a bank is defined (minimum size, relief, etc.), and there are likely many more sites with topographical relief sprinkled throughout the continental shelf off South Texas. Within the Mysterious Banks complex alone, Bright and Rezak (1976) identified 28 distinct reefs through direct observation from a submersible. Seismic studies reveal that there are about 40 outer–shelf banks of coralgal origin and that there were likely other carbonate reefs (possibly oyster banks) during the Wisconsinan period that have since been buried by more recent deposits (Berryhill 1986). Holocene deposits have buried only the edges of many Wisconsinan carbonate reefs, leaving the upper portions exposed as hard–substrate banks (Berryhill 1986), which we focus on in this paper. The existing banks are in about 60–95 m (Rezak et al. 1985, Berryhill 1986) of water with the banks typically cresting at 55–82 m (Rezak et al. 1985, Belopolsky and Droxler 1999) and having an average relief of 10–12 m (Bright and Rezak 1976). Most sites have only minor encrusting populations of coralline algae with a mix of species similar to those of the algal–sponge and drowned reef zones of the Flower Garden Banks, and the epifaunal diversity is minimal below 75 m (Rezak et al. 1985, Dennis and Bright 1988). Although most South Texas Banks have low vertical relief, they are still conspicuous on the vast, flat continental shelf. The banks are located in a mid–to–high–energy environment, thus experiencing high turbidity and substantial sedimentation. The larger South Texas Banks are located roughly on the outer South Texas continental shelf, approximately in the area where freshwater inputs transition from being influenced by Texas rivers to being dominated by the Mississippi River (Flint and Rabalais 1981). Rezak et al. (1985) delineated the specific depth ranges of biotic zones at several sites; biotic zones at the South Texas Banks include an antipatharian transitional zone, a nepheloid layer, and soft bottom. Additionally, Rezak et al. (1985) created profiles of observed biota according to zones at some of the South Texas Banks. The dynamic nepheloid layer is composed of resuspended sediment that is subject to variable, yet chronic, turbidity. The nepheloid layer heavily influences the ecology of the South Texas Banks (Tunnell et al. 2009).

In this analysis species richness at Southern Bank was analyzed specifically because the most data are available for that site, which is likely because of the geological complexity of the site when compared to the other South Texas Banks. Southern Bank is located about 55 km offshore of Corpus Christi on the edge of the outer continental shelf (Belopolsky

TABLE 1. *Known locations and other data for 21 outer–shelf South Texas Banks (ordered from north to south). Sources[1]: Rezak et al. 1985, Berryhill 1986, Bright and Rezak 1976. N/A: Data not available.*

Bank Name	Latitude	Longitude	Depth to Base (m)	Depth to Crest (m)	Relief (m)	Area (km²)
Bad Mud Bank	27°52.50'N	96°07.0'W	58	N/A	N/A	0.65
Small Dunn Bar	27°49.0'N	96°07.6'W	65	N/A	N/A	0.65
Big Dunn Bar	27°49'N	96°08.4'W	65	N/A	N/A	1.20
Baker Bank	27°45.0'N	96°14.0'W	74	58	16	2.09
East Breaks	27°43.0'N	95°41.0'W	100	N/A	N/A	15.5
South Baker Bank	27°40.5'N	96°16.4'W	82	64	18	N/A
Aransas Bank	27°35.5'N	96°27.0'W	70	58	12	0.67
North Hospital Bank	27°34.5'N	96°28.5'W	68	58	10	1.60
Hospital Bank	27°32.5'N	96°28.5'W	78	56	22	5.22
Southern Bank	27°26.5'N	96°31.5'W	80	60	20	1.27
Little Mitch Bank	27°18.0'N	97°10.0'W	30	N/A	N/A	2.5
Four Leaf Clover	27°09.07'N	97°10.01'W	30	N/A	N/A	2.5
Dream Bank	27°02.5'N	96°42.5'W	84	70	14	2.95
9 Fathom Rocks[2]	26°57.5'N	97°18.51'W	20	N/A	N/A	2.5
Big Adam Rock	26°57.2'N	96°49.0'W	66	60	6	0.37
Small Adam Rock	26°56.7'N	96°49.8'W	62	60	2	0.04
Blackfish Ridge	26°52.6'N	96°46.6'W	72	60	12	1.12
Mysterious Bank[3]	26°46.1'N	96°42.0'W	74 – 84	N/A	N/A	N/A
Steamer Bank	26°49.3'N	97°13.50'W	30	N/A	N/A	N/A
Seabree Banks[4]	26°24.3'N	96°58.3'W	36	33	3	130
East Bank	26°02.78'N	96°54.6'W	40	N/A	N/A	155.4

[1] Additional source: Circé, R.C. and J.W. Tunnell, Jr. 1987. Distribution of shelf and shelf–edge banks in the northwestern Gulf of Mexico. Abstract and poster, Fourth Annual Mid–Year SEPM Meeting, Austin, TX, USA.
[2] 9 Fathom Rocks is a complex consisting of at least three sites.
[3] Mysterious Bank is a complex (6.4 km by 2.7 km) of small (100 m) to medium (1250 m) banks.
[4] Seabree (also Sebree) Bank(s) includes four sites.

and Droxler 1999). The formation is about 40–50 m thick but has a maximum relief of 22m above the seafloor (Bright and Rezak 1976, Belopolsky and Droxler 1999). There are three to four levels of reef development or die–offs, which are similar to more defined terraces on sites in the diapiric area in the northwestern GOM, and Southern Bank as it exists today has two distinct peaks that represent the reef crests. These terraces at Southern Bank may represent successive die–offs that occurred during the late Holocene (Holcombe et al. 2010, Belopolsky and Droxler 1999).

Results

Data were reviewed for 21 known outer–shelf South Texas Banks (Table 1). Compared to more northern Gulf sites, such as the East and West Flower Garden Banks that have been well studied over time, few studies have been conducted on the South Texas Banks, and most of those were conducted in the 1970s and 1980s (Table 2). These studies were primarily faunal inventories intended to provide baseline data prior to oil and gas development and often did not include any information of faunal seasonality. No robust dataset quantifies biodiversity at the South Texas Banks comprehensively, but analysis of observational, anecdotal, and scientific data from existing reports, published and unpublished, provides

ecological insight regarding community types and possible connectivity throughout the GOM region.

Collectively, 28 references describe observations of about 1,282 unique taxa at the nearshore 7.5 Fathom Reef and 11 of the outer–shelf South Texas Banks (Tables 2 and 3). This total accounts for taxa identified at more than one site. Of the 11 outer–shelf sites, Seabree Bank is the only site that is a relict barrier relief; the remaining 10 sites are relict coralgal reefs. Of all these banks, 7.5 Fathom Reef, Southern Bank, and Hospital Rock each host roughly a third of the total taxa known from the literature. Based on available data describing communities at the South Texas Banks, about 75% of the 454 taxa identified as occurring at 7.5 Fathom Reef were documented exclusively at 7.5 Fathom Reef, which is the site nearest to shore in this study. Therefore, only 115 taxa recorded at 7.5 Fathom Reef have been observed in both nearshore and offshore environments. The outer–shelf South Texas Banks (mostly coralgal banks) collectively provide habitat for about 943 known taxa, based on currently available information. Of these taxa, about 27% do not have occurrences from specific sites mentioned.

Only slightly over half of the identified taxa were identified to species level. Not unexpectedly, outer–shelf banks surveyed more often tend to have the highest species

TABLE 2. *References containing taxonomic data for South Texas Banks ordered by date.*

Citation	Source Type	Site(s)	Taxa
Parker and Curray 1956	Peer–reviewed journal	Baker Bank, Southern Bank	Cnidaria, Mollusca
Hoese 1965	PhD dissertation	Unspecified South Texas Banks	Chordata: Chondrichthyes, Actinopterygii
Causey 1969	Master's thesis	7.5 Fathom Reef	Chordata: Chondrichthyes, Actinopterygii
Tunnell 1969	Master's thesis	7.5 Fathom Reef	Mollusca
Tunnell and Chaney 1970	Peer–reviewed journal	7.5 Fathom Reef	Mollusca
Felder 1971	Master's thesis	7.5 Fathom Reef	Annelida, Arthropoda
Tunnell 1972	Peer–reviewed journal	7.5 Fathom Reef	Brachiopoda
Bright and Pequegnat 1974	Book	Seabree Bank	Granuloreticulosa, Cnidaria, Mollusca, Annelida, Arthropoda, Echinodermata, Chordata
McCarty 1974	Master's thesis	7.5 Fathom Reef	Annelida: Polychaeta
Shirley 1974	Master's thesis	7.5 Fathom Reef	Cnidaria, Sipuncula, Echinodermata
Abbott and Bright 1975	Technical report	Individual species reports from Small Adam Rock, Big Adam Rock, Blackfish Ridge, Mysterious Bank, Southern Bank, Baker Bank, 7.5 Fathom Reef	Porifera, Cnidaria, Mollusca, Bryozoa, Chordata, Annelida, Arthropoda, Brachiopoda, Echinodermata
Bright and Rezak 1975	Technical report	Unspecified South Texas Banks, Southern Bank	Chordata: Perciformes, Anguilliformes, Tetraodontiformes
Davis 1975	Master's thesis	7.5 Fathom Reef	1 species: *Leptogorgia virgulata*
Bright and Rezak 1976	Government report	Baker Bank, Big Adam Rock, Dream Bank, Hospital Rock, North Hospital Rock, Seabree Bank, South Baker Bank, Southern Bank, 7.5 Fathom Reef, unspecified South Texas Banks	Granuloreticulosa, Rhodophyta, Chlorophyta, Ochrophyta, Porifera, Cnidaria, Mollusca, Annelida, Sipuncula, Arthropoda, Chordata, Brachiopoda, Echinodermata
Groover et al. 1977	Government report	Hospital Rock, Southern Bank	Annelida: Polychaeta
Holland 1976	Government report	Hospital Rock, Southern Bank	Mollusca, Annelida, Arthropoda
UTMSI 1976	Government report	Hospital Rock, Southern Bank	Cnidaria; Mollusca; Annelida; Arthropoda: Ostracoda, Malacostraca; Echinodermata
Rezak and Bright 1978	Government report	Unspecified South Texas Banks	Cnidaria, Chordata
Felder and Chaney 1979	Peer–reviewed journal	7.5 Fathom Reef	Annelida, Arthropoda
Tunnell 1982	Symposium proceedings	7.5 Fathom Reef	Brachiopoda
Rezak et al. 1985	Book	Baker Bank, Big Adam Rock, Blackfish Ridge, Mysterious Bank, Seabree Bank, Small Adam Rock, Southern Bank, 7.5 Fathom Reef, unspecified South Texas Banks	Rhodophyta, Chlorophyta, Ochrophyta, Porifera, Cnidaria, Mollusca, Annelida, Arthropoda, Brachiopoda, Bryozoa, Echinodermata, Chordata
Dennis and Bright 1988	Peer–reviewed journal	Big Adam Rock, Blackfish Ridge, Mysterious Bank, Small Adam Rock, Southern Bank, unspecified South Texas Banks	Chordata: Chondrichthyes, Actinopterygii
Rezak et al. 1990	Peer–reviewed journal	Southern Bank	Porifera, Mollusca, Arthropoda, Brachiopoda, Chordata
Dokken et al. 1993	White paper	Seabree Bank	Cnidaria, Mollusca, Annelida, Arthropoda, Echinodermata
Lehman et al. 1996	Government report	7.5 Fathom Reef	Rhodophyta, Chlorophyta, Chordata, Ochrophyta, Cnidaria, Mollusca, Annelida, Arthropoda, Echinodermata
Hyde 2000	Master's thesis	Hospital Rock, 7.5 Fathom Reef	Mollusca: Gastropoda, Bivalvia, Scaphopoda
Weaver et al. 2005	Government report	7.5 Fathom Reef	Chordata: Chondrichthyes, Actinopterygii
Weaver et al. 2009	Symposium proceedings	Southern Bank, North Hospital Bank	Chordata: Actinopterygii

TABLE 3. Number of taxa at selected South Texas Banks. Sites are listed north to south. No taxonomic data were available for the South Texas Banks not included in this table.

Bank Name	Taxa[1]		Species	
	Identified	Unique to Bank	Identified	Unique to Bank
Baker Bank	48	2	27	0
South Baker Bank	43	5	18	0
North Hospital Bank	44	2	28	0
Hospital Bank	363	125	239	67
Southern Bank	420	96	268	49
Dream Bank	62	13	27	0
Big Adam Rock	14	0	5	0
Small Adam Rock	6	0	3	0
Blackfish Ridge	6	0	3	0
Mysterious Bank	6	0	3	0
Seabree Bank	42	19	34	9
Collective banks (sites unspecified in literature)	389	253	250	157
7.5 Fathom Reef	454	339	417	302
ALL BANKS COMBINED[2]	**1282**		**881**	

[1] Taxa include organisms identified to any level (species or above species level).

[2] These totals account for overlap of counts among banks; each taxon is counted only once in sums.

richness. Southern and Hospital Banks are the most speciose of the studied offshore South Texas Banks, even though fewer studies focused on Hospital Bank than Southern Bank (Figure 3).

The number of taxa listed in each classification category increases as taxonomic level decreases. Taxa representing 14 phyla have been recorded from the South Texas Banks, and most of these taxa are placed in the phyla Annelida (27%), Mollusca (23%), and Chordata (17%) (Figure 4). Within Annelida, all species identified are within Class Polychaeta.

When looking at Southern Bank specifically as a case study, the top three phyla are Annelida, Chordata, and Mollusca, as seen with the South Texas Banks as a group. However, at Southern Bank there are no reports of Bryozoa, Granuloreticulosa, Rhodophyta, or Sipuncula, which occur at one or more of the other South Texas Banks. This may be due to the type of sampling at this location rather than to the absence of these taxa. Of the 269 species reported at Southern Bank, almost 49% are classified within the Phylum Annelida. Sixteen percent of the species at Southern Bank are in the Order Phyllodocida (polychaetes), and 12% are in Perciformes (bony fishes) (Figure 5). The remaining orders individually com-

prise less than 10% of the species recorded at Southern Bank. There are 17 polychaete species that are not assigned to orders in the Biodiversity of the Gulf of Mexico Database (Moretzsohn et al. 2013) and other online taxonomic databases (World Register of Marine Species, Encyclopedia of Life, Integrated Taxonomic Information System) due to recent phylogenies debunking older classification schemes (Rouse and Fauchald 1997). These unclassified polychaete species make up about 6% of the species listed at Southern Bank.

Discussion

Southern and Hospital Banks show the highest species richness of the outer–shelf South Texas Banks for several reasons. From the biophysical perspective, Hospital and Southern Banks are large sites with the highest vertical relief of the outer–shelf South Texas Banks (see Table 1). Greater topographic relief is conducive to increased ecological zonation, which provides more habitat, not only based on surface area and vertical structure but also on various physical parameters such as depth, turbidity, and light penetration (Rezak et al. 1985). Different species inhabit different biotic zones; therefore, sites with greater vertical relief have higher spe-
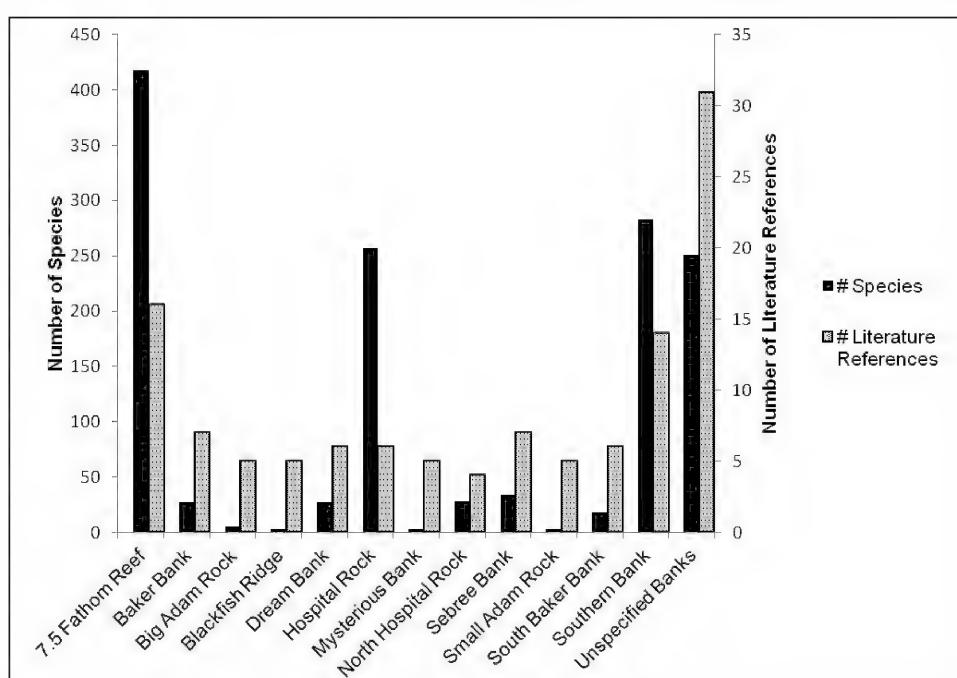
**Figure 3.** Number of species and literature references per bank.
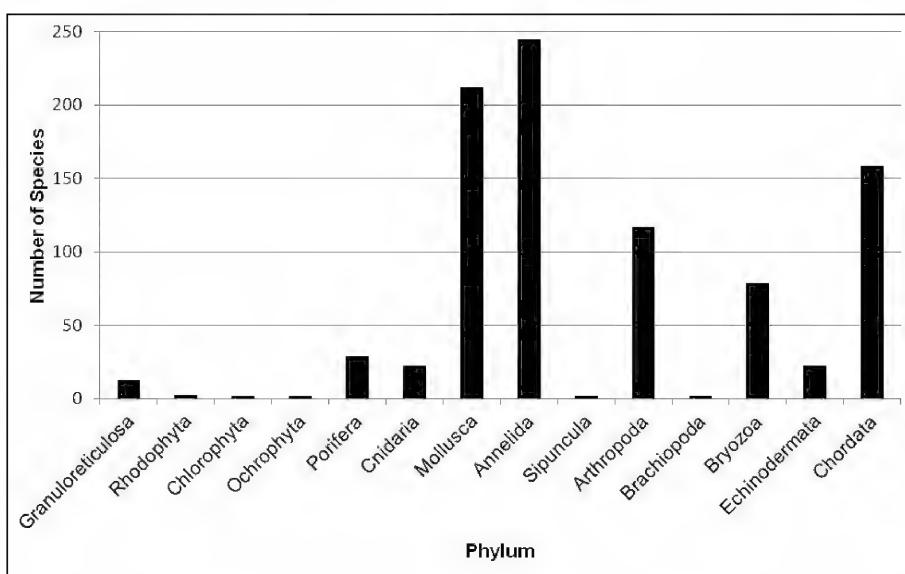
Figure 4. *Number of species per phylum at South Texas Banks.*

cies richness. Also, the spatial and temporal variability of the thickness and density of the nepheloid layer may enhance or impede observations at the time of data collection (Flint and Rabalais 1981). As topographical highs, Southern and Hospital Banks protrude farther above the nepheloid layer than the other banks, enabling different ecological communities to survive (Rezak et al. 1985). The nepheloid layer's coverage of other banks with lower relief is likely a reason they are not as well characterized because the suspended muddy sediment obscures visibility required for biological observations. From the perspective of scientific effort, higher study effort is often correlated with higher species richness because more time has been spent looking for organisms, which increases the probability of finding more species. Higher species richness at several banks could be due to a larger number of studies at such banks (see Figure 3), but this is only a rough measure of sampling effort because of differences in effort among studies. Many studies identified taxa from the South Texas Banks collectively; as expected, such studies revealed relatively high biodiversity as well. In the case of the outer–shelf South Texas Banks, greatest scientific effort at Southern Bank is likely due to accessibility, size, and ease of locating the site. Similar reasons apply to the high species richness identified at the nearshore 7.5 Fathom Reef; more studies occurred at this reef than at any other site discussed herein. Data are no doubt lacking for many of the South Texas Banks because exact locations of sites are not always known, some sites have more than one name causing confusion when trying to match data with sites in literature, atmospheric and sea conditions may make it difficult to locate a site, and the culture of secrecy within the fishing communities makes it difficult for scientists to locate or study sites known to have a high abundance of targeted species.

Although sampling effort biases may exist, Southern Bank appears to have the greatest species richness when compared to other South Texas Banks, therefore allowing further analysis. The pattern of diversity of the higher taxonomic groups remains similar to the combined South Texas Banks, with Annelida, Chordata, and Mollusca being the most speciose phyla. The number of species from Southern Bank, however, does not skew the combined data. When Southern Bank was removed from the combined South Texas Banks data, the percentage of species changed by one percentage point or less for every phylum recorded. This indicates that Southern Bank could be representative of the potential species diversity that may be found at other South Texas Banks.

In previous studies of the submerged banks of the northwestern GOM, mollusks were the most diverse group of organisms (Tunnell et al. 1978). Although results from this study show that polychaetes represent the taxonomic group with the highest species richness, mollusks have the second highest species richness among the South Texas Banks. However, biases in data collection methods and analyses may obfuscate or even invalidate some conclusions. Potential biases include, but are not limited to:

- Taxonomic misidentifications
- Unresolved or inconsistent taxonomic shifts in the last several decades
- Variable sampling methods (consistency, duration, gear type, etc.) and effort among sites
- Reliability of anecdotal and observational data (e.g., from submersibles)
- Differences in data interpretation by different scientists
- Lack of repeatability
- Lack of geographic coordinates in reports and site nomenclature confusion
- Spatiotemporal variability of nepheloid layer.

Comparisons within taxonomic groups (corals, mollusks, fishes, etc.) may provide additional insight into community structure and potential connectivity among the banks. For example, previous qualitative comparisons indicate that the reef fish community of the South Texas Banks resembles that of some banks in the salt diapir area to the north at similar depths (Weaver et al. 2009). Further analyses could focus on numbers of species that are typical of temperate and tropical marine communities to determine the role of the South Texas Banks in providing habitat essential for the GOM to function as a faunal ecotone (Bangma and Haedrich 2008). The overall morphological similarity of the South Texas Banks to the Lobos–Tuxpan Reef System in Mexico (Weaver et al.
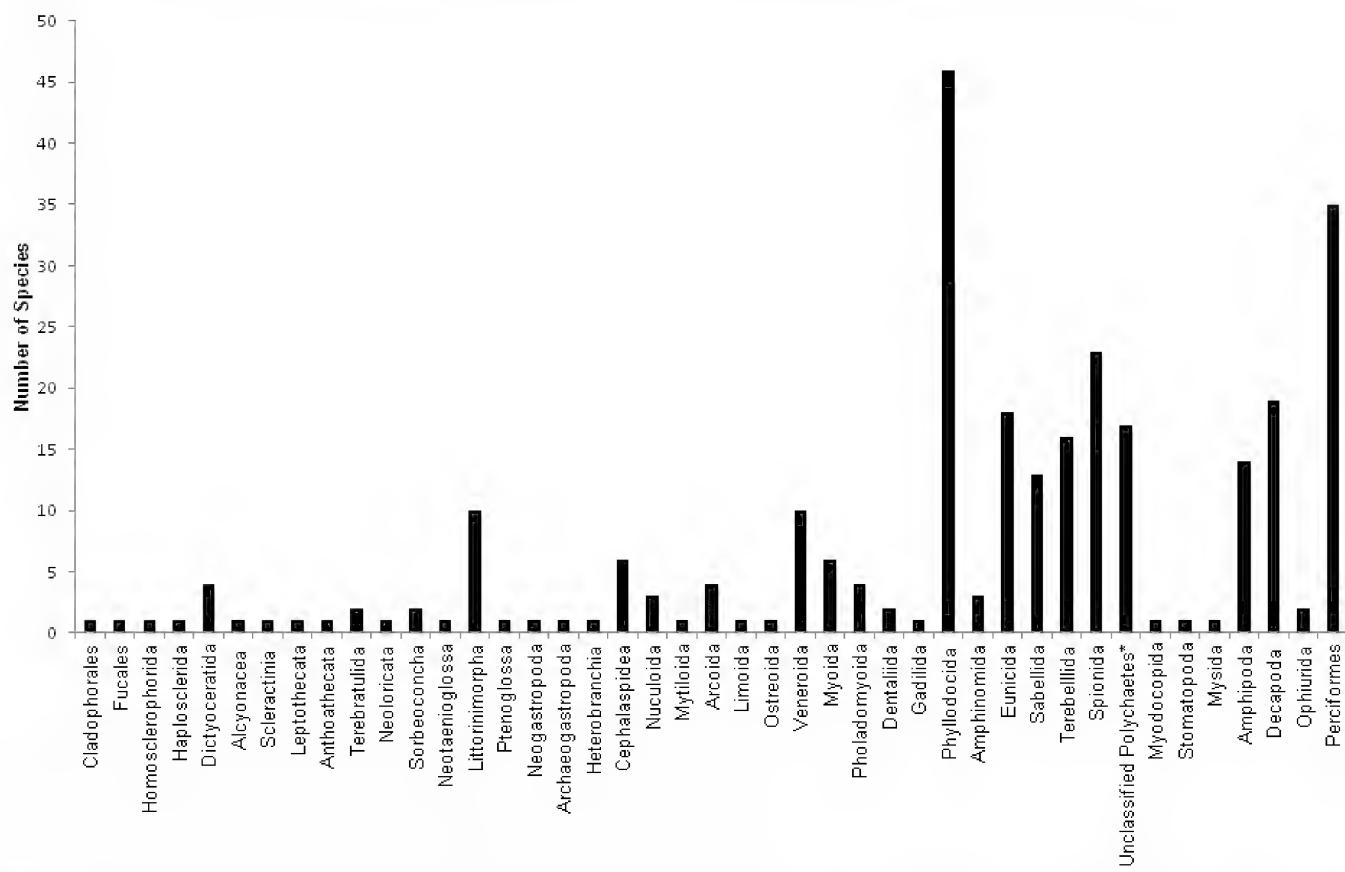
Figure 5. *Number of species per order at Southern Bank. Recent large–scale phylogenies of polychaetes have found that in older classifications some of the orders are not monophyletic (Rouse and Fauchald 1997). Therefore, recent classification schemes have not assigned some polychaete families to any order; these are referred to as unclassified polychaetes.*

2009) may indicate that Tuxpan biological studies (Rigby and McIntire 1966, Chávez et al. 1970, Tunnell 1974, Tunnell et al. 2007, Escobar–Vásquez and Chávez 2012) could serve as good models for future studies at the South Texas Banks given that the two systems likely share ecological connections as a result of their relatively close proximity to each other and to currents. Increases in species richness, rank abundance, and occurrence of tropical fish species since the 1970s (Bright and Rezak 1976, Tunnell et al. 2009) supports the concept of ecological connectivity, perhaps between the Lobos–Tuxpan Reef System and the South Texas Banks, particularly in a changing climate.

Information about the South Texas Banks is dominated by physical and geological data, with relatively little biological data being available. Because there have not been systematic, comprehensive studies conducted on all of the South Texas Banks, little is known about the ecological communities at many of the sites. Suggested next steps include systematic ecological studies using a consistent methodology for describing biota of all South Texas Banks, standardization of specific locations and names of sites (see note in Table 1), and discussion of species richness as functions of survey effort, habitat availability, and environmental conditions. These studies would facilitate design of conservation and protection parameters for rare habitat sites with vertical relief found in some portions of the Gulf's continental shelf. Surveys and observations would likely yield the best results during May to November when water clarity is greatest due to northerly currents moving clear, tropical oceanic water from the south over the banks, thus reducing the visual obscurity of the nepheloid layer (Tunnell et al. 2009). Finally, survey results could be compared to similar studies conducted on hard banks and reefs throughout the rest of the GOM to draw conclusions and propose candidate sites for protection on a regional scale (Nash and McLaughlin 2012). We submit that such studies are particularly important for habitat conservation and sustainability of living marine resources in the face of climate change.

Literature Cited

Abbott, R.E. and T.J. Bright. 1975. Benthic communities associated with natural gas seeps on carbonate banks in the northwestern Gulf of Mexico. Report for Study of Naturally Occurring Hydrocarbons in the Gulf of Mexico. Texas A&M University Press, College Station, TX, USA, 191 p.

Bangma J.L. and R.L. Haedrich. 2008. Distinctiveness of the mesopelagic fish fauna in the Gulf of Mexico. Deep Sea Research II 55:2508–2596.

Belopolsky, A.V. and A.W. Droxler. 1999. Uppermost Pleistocene transgressive coralgal reefs on the edge of the South Texas shelf: analogs for reefal reservoirs buried in siliciclastic shelves. Proceedings of the Gulf Coast Section Society for Sedimentary Geology Foundation 19th Annual Research Conference Advanced Reservoir Characterization, 5–8 December 1999, p. 41–50.

Berryhill, H.L., Jr. 1986. Late Quaternary facies and structure, northern Gulf of Mexico: interpretations from seismic data. The American Association of Petroleum Geologists, Tulsa, OK, USA, 289 p.

Bright, T.J. and L.H. Pequegnat, eds. 1974. Biota of the West Flower Garden Bank. Gulf Publishing Company, Houston, TX, USA, 435 p.

Bright, T.J. and R. Rezak. 1975. Fishing banks of the Texas continental shelf. Technical Report. Department of Oceanography, Texas A&M University, College Station, TX, USA, 45 p.

Bright, T.J. and R. Rezak. 1976. A biological and geological reconnaissance of selected topographical features of the Texas continental shelf. Contract 08550–CT5–4. Final Report. U.S. Department of Interior, Bureau of Land Management, Outer Continental Shelf Office, New Orleans, LA, USA. 377 p.

Carsey, J.B. 1950. Geology of Gulf coastal area and continental shelf. Bulletin of American Association of Petroleum Geologists 34:361–385.

Causey, B.D. 1969. The fish of Seven and One–Half Fathom Reef. Master's thesis. Texas A&I University, Kingsville, TX, USA, 110 p.

Chávez, E.A., E. Hidalgo, and M.L. Sevilla. 1970. Datos acerca de las comunidades bentónicas del arrecife de Lobos, Veracruz. Revista de la Sociedad Mexicana de Historia Natural 31:211–280.

Davis, J.K. 1975. Factors influencing the presence of red snapper (*Lutjanus campechanus* Poey) on Seven and One–Half Fathom Reef. Master's Thesis. Texas A&I University, Kingsville, TX, USA, 124 p.

Dennis, G.D. and T.J. Bright. 1988. Reef fish assemblages on hard banks in the northwestern Gulf of Mexico. Bulletin of Marine Science 43:280–307.

Dokken, Q., R.L. Lehman, J. Prouty, C. Adams, and C. Beaver. 1993. A preliminary survey of Sebree Bank (Gulf of Mexico, Port Mansfield, Texas) August 23–27, 1993. Technical Report 9305. Center for Coastal Studies, Texas A&M University–Corpus Christi, Corpus Christi, TX, USA, 13 p.

Earle, S.A. 1998. Foreward–Dedicated Issue: Flower Garden Banks National Marine Sanctuary. Gulf of Mexico Science 16:127.

Escobar–Vásquez, C. and E.A. Chávez. 2012. Coral community structure at Isla Lobos reef, Gulf of Mexico. Proceedings, Twelfth International Coral Reef Symposium, Cairns, Australia, July 9–13, 4 p.

Felder, D.L. 1971. The decapod Crustacea of Seven and One–Half Fathom Reef. Master's thesis. Texas A&I University, Kingsville, TX, USA, 101 p.

Felder, D.L. and A.H. Chaney. 1979. Decapod crustacean fauna of Seven and One–Half Fathom Reef, Texas: species composition, abundance, and species diversity. Contributions in Marine Science 22:1–29.

Flint, R.W. and N.N. Rabalais. 1981. Environmental Studies of a Marine Ecosystem. University of Texas Press, Austin, TX, USA, 240 p.

Geodicke, T.R. 1955. Origin of the pinnacles on the continental shelf and slope of the Gulf of Mexico. Texas Journal of Science 7:149–159.

Groover, R.D., G.P. Pfeiffer, C.W. Griffin, D.A. Kalke, and T.C. Moore. 1977. Environmental studies, South Texas outer continental shelf, biology and chemistry. Contract AASS0–CT6–17. Final Report. Bureau of Land Management, Washington, DC, USA. Volume I, Chapters 1 – 8, 564 p.

Hoese, H.D. 1965. Spawning of marine fishes in the Port Aransas, Texas area as determined by the distribution of young and larvae. Ph.D. dissertation. University of Texas, Austin, TX, USA, 144 p.

Holcombe, T.L., C.A. Arias, and W.R. Bryant. 2010. Northwestern Gulf of Mexico bathymetry and geomorphology. In: J. Breman (ed.). Ocean Globe. ESRI Press Academic, Redlands, CA, USA, p. 135–178.

Holland, J.S. 1976. Environmental studies, South Texas outer continental shelf, biology and chemistry. Contract AA550–CT6–17. Final Report. Bureau of Land Management, Washington, DC, USA. Volume II, Chapters 9 – 18, 710 p.

Hyde, L.J. 2000. The systematics, distribution, and ecology of the mollusks of Stetson Bank, northwestern Gulf of Mexico. Master's thesis, Texas A&M University, Corpus Christi, TX, USA, 216 p.

Lehman, R.L, C. Beaver, S. Cox, K. Hancock, P. Choucair, and A. Oswald. 1996. A twenty–five year resurvey of Seven and One–Half Fathom Reef (Gulf of Mexico, Padre Island, Texas). Final Report. Minerals Management Service, Washington, DC, USA, 20 p.

McCarty, D.M. 1974. The polychaetes of Seven and One–Half Fathom Reef. Master's thesis. Texas A&I University, Kingsville, TX, USA, 104 p.

Moretzsohn, F., J.A. Sanchez Chavez, J.W. Tunnell, Jr., editors. 2013. GulfBase: Resource Database for Gulf of Mexico Research. http://www.gulfbase.org. (viewed on 1/7/2013).

Nash, H.L. and R.J. McLaughlin. 2012. Opportunities for trinational governance of ecologically connected habitat sites in the Gulf of Mexico. KMI International Journal of Maritime Affairs and Fisheries 4:1–32.

Parker, R.H. and J.R. Curray. 1956. Fauna and bathymetry of banks on continental shelf, northwest Gulf of Mexico. Bulletin of the American Association of Petroleum Geologists 40:2428–2439.

Rezak, R. and T.J. Bright. 1978. South Texas topographic features study. Contract No. AA550–CT6–18. Executive summary of the Final Report. US Department of the Interior Bureau of Land Management, Outer Continental Shelf Office, New Orleans, LA, USA, 60 p.

Rezak, R., T.J. Bright, and D.W. McGrail. 1985. Reefs and Banks of the Northwestern Gulf of Mexico: Their Geological, Biological, and Physical Dynamics. John Wiley & Sons, Inc., New York, NY, USA, 259 p.

Rezak, R., S.R. Gittings, and T.J. Bright. 1990. Biotic assemblages and ecological controls on reefs and banks of the northwest Gulf of Mexico. American Zoologist 30:23–35.

Rigby, J.K. and W.G. McIntire. 1966. The Isla de Lobos and associated reefs, Veracruz, Mexico. Brigham Young University Geological Studies 13:3–46.

Roberts, H.H. 2011. Surficial geology of the northern Gulf of Mexico continental slope. Impacts of fluid and gas expulsion. In: N.A. Buster and C.W. Holmes (eds.). Gulf of Mexico Origin, Waters, and Biota. Volume 3, Geology. Texas A&M University Press, College Station, TX, USA, p. 209–228.

Rouse, G. and K. Fauchald. 1997. Cladistics and polychaetes. Zoologica Scripta 26:139–204.

Shirley, T.C. 1974. The echinoderms of Seven and One–Half Fathom Reef. Master's Thesis. Texas A&I University, Kingsville, TX, USA, 82 p.

Thayer, P.A., A. La Rocque, and J.W. Tunnell, Jr. 1974. Relict lacustrine sediments on the inner continental shelf, Southeast Texas. Transactions of the Gulf Coast Association of Geological Societies 24:337–347.

Tunnell, J.W., Jr. 1969. The mollusks of Seven and One–Half Fathom Reef. Master's Thesis. Texas A&I University, Kingsville, TX, USA, 83 p.

Tunnell, J.W., Jr. 1972. *Crania* sp. (Brachiopoda) from Texas waters. The Texas Journal of Science 23:553.

Tunnell, J.W., Jr. 1974. Ecological and geographical distribution of Mollusca of Lobos and Enmedio coral reefs, Southwestern Gulf of Mexico. Ph.D. Dissertation. Texas A&M University, College Station, TX, USA, 158 p.

Tunnell, J.W., Jr. 1982. Distribution and habitat of *Discradisca antillarum* (d'Orbigny, 1846) (Brachiopoda: Inarticulata) in the western Gulf of Mexico. Proceedings, South Texas Fauna: A Symposium Honoring Dr. Allan H. Chaney, Caesar Kleberg Wildlife Research Institute, Kingsville, TX, USA, p. 37–47.

Tunnell, J.W., Jr. and B.D. Causey. 1969. Vertebrate Pleistocene fossils from the continental shelf, northwestern Gulf of Mexico. Texas A&I University Studies 2:75–76.

Tunnell, J.W., Jr. and A.H. Chaney. 1970. A checklist of the mollusks of Seven and One–Half Fathom Reef, northwestern Gulf of Mexico. Contributions in Marine Science 15:193–203.

Tunnell, J.W., Jr., S.A. Earle, and R.R. Furgason. 2005. Foreward–Dedicated Issue: Flower Garden Banks National Marine Sanctuary. Gulf of Mexico Science 23:1.

Tunnell, J.W., Jr., E.A. Chávez, and K. Withers. 2007. Coral Reefs of the Southern Gulf of Mexico. Texas A&M University Press, College Station, TX, USA, 194 p.

Tunnell, J.W., Jr., D.C. Weaver, and T.C. Shirley. 2009. Recent research on South Texas topographic features: ecology. Proceedings, Twenty–fifth Gulf of Mexico Information Transfer Meeting, M. McKay and J. Nides (eds.). OCS Study MMS 2009–051, New Orleans, LA, USA, January 2009, p. 202–209.

Tunnell, J.W., Jr., J.C. Woods, M.E. Kindinger, and J.L. Kindinger. 1978. Fauna of shelf–edge submarine banks in the northwestern Gulf of Mexico. Contract 14–08–0001–G–381. Final Report. U.S. Geological Survey, Office of Marine Geology, Washington, DC, USA, 66 p.

UTMSI (University of Texas Marine Science Institute). 1976. Environmental studies, South Texas outer continental shelf, biology and chemistry. Contract AA550–CT6–17. Final Report, Volume VI, Appendices E,F,G,H,I,J,K. Bureau of Land Management, Washington, DC, USA, 1014 p.

Weaver, D., E. Hickerson, and G. Schmahl. 2005. Deep reef fish surveys by submersible on Alderdice, McGrail, and Sonnier Banks in the Northwestern Gulf of Mexico. Flower Garden Banks National Marine Sanctuary, NOAA, Galveston, TX, USA, p. 69–87.

Weaver, D.C., J.W. Tunnell, Jr., and T.C. Shirley. 2009. Recent research on South Texas topographic features: mapping. Proceedings, Twenty–fifth Gulf of Mexico Information Transfer Meeting, M. McKay and J. Nides (eds.). OCS Study MMS 2009–051, New Orleans, LA, USA, January 2009, p. 193–201.

Assessment of Seagrass Floral Community Structure from Two Caribbean Marine Protected Areas

Paul A.X. Bologna
Montclair State University

Anthony J. Suleski
Montclair State University

ASSESSMENT OF SEAGRASS FLORAL COMMUNITY STRUCTURE FROM TWO CARIBBEAN MARINE PROTECTED AREAS

Paul A. X. Bologna[1*] and Anthony J. Suleski[2]

[1]Department of Biology and Molecular Biology, Montclair State University, 1 Normal Ave., Montclair, NJ 07043 USA; [2]Aquatic and Coastal Sciences Program, Montclair State University, 1 Normal Ave, Montclair, NJ 07043 USA; *Corresponding author, email: Bolognap@mail.montclair.edu

ABSTRACT: Seagrass communities represent spatially complex and biomass producing systems comprised of intermixed seagrass and algal species. We investigated shallow water communities from two Marine Protected Areas (MPAs) in the Caribbean: St. John, United States Virgin Islands and Cayos Cochinos, Honduras. St. John sites (4) lie within the Virgin Islands National Park and the Coral Reef National Monument and are designated within an UNESCO Biosphere Reserve. Honduran sites (4) lie within the designated Marine National Monument. Our results indicate that both MPAs were dominated by *Thalassia testudinum* with spatial coverage and shoot density significantly greater in Honduras. Many sites also showed substantial cover of *Syringodium filiforme*, which was significantly greater in St. John. Most major algal groups showed significant differences between MPAs and among sites within locations. Specifically, *Halimeda, Penicillus, Udotea, Galaxaura,* and *Dictyosphaeria* were significantly more abundant in Honduras, while *Padina* and *Avrainvillea* were significantly greater from St. John. Additionally, only Honduran sites showed the presence of coral colonies (*Montastrea* and *Porites*) within their seagrass beds. Floral community level analyses demonstrated significant differences among almost all site comparisons suggesting relatively distinct floral communities exist within each of these regions, but both MPAs maintain high spatial coverage of seagrasses providing critical ecosystem services.

KEY WORDS: Cayos Cochinos, Honduras, MPA, *Syringodium, Thalassia*

INTRODUCTION

Global seagrass decline is well documented (Orth et al. 2006, Waycott et al. 2009) and is often a result of anthropogenic activities. While seagrasses are recognized as important primary producers and serve as refuge from predation, our ability to protect and stop the decline is limited for many species (Short et al. 2011). Consistently, coastal eutrophication has been a significant factor in seagrass decline as excess nitrogen fuels phytoplankton and ephemeral algal populations, causing reductions in light and diurnal hypoxia as blooms spread. While this scenario is well documented in temperate systems, the decline in some tropical grass beds is more difficult to link directly to eutrophication. Often tropical seagrass beds have shown declines associated with physical damage (e.g., boat anchors, mooring, prop scars), light attenuation from terrestrial sediment runoff and from the generalized collapse and destruction of the associated mangrove and coral reef communities. Both mangrove and coral reef systems have also seen significant global declines from a variety of direct anthropogenic sources (e.g., coastal development, aquaculture, run–off, eutrophication, algal overgrowth) as well as widespread diseases, overfishing, loss of herbivores, and climate change (Valiela et al. 2001, Pandolfi et al. 2003). Essentially, when any of these communities is negatively impacted, secondary impacts are often felt in the adjacent systems. For example, mangrove deforestation leads to elevated run–off (Thampanya et al. 2006) and resuspension causing sediment loading to seagrass and coral reef communities; thereby increasing light stress and physical smothering of organisms. These secondary ripple effects eventually lead to overall declines in entire coastal ecosystems and a

vicious circle of ever increasing declines occur. Rivera–Monroy et al. (2004) describe a framework in which to assess the current status of the Caribbean marine systems and identify potential stressors and areas of research necessary to develop management strategies. Wilkinson and Salvat (2012) argue that this unprecedented decline rises to the moral imperative of addressing social issues driving the anthropogenic stresses negatively impacting habitats and living resources, while Unsworth and Cullen (2010) advocate for the conservation of seagrasses as fundamental to the health of tropical marine communities.

Habitat restoration is often proposed as a mechanism to reverse the trends of habitat loss, but it has met with varying levels of success, and restoration of mangrove and coral reef communities may take decades to become established and provide biological and ecosystem services (Bosire et al. 2008). As such, one potential avenue to stem the local declines in these systems is to enact policies which protect them from further decay and allow the natural resilience of the communities to reestablish and flourish. The most frequent policy employed is the development of a Marine Protected Area (MPA). While this may seem to be a relatively straightforward answer, MPA implementation often means the displacement of local residents and elimination of their livelihood. This is most evident in the implementation of fishing restrictions, which may impact both local economies and basic food gathering for the affected, displaced individuals (Brondo and Bown 2011). As such, development of MPAs frequently takes substantial negotiations to ensure local economic viability or subsistence, but local communities can

often find themselves marginalized with limited input, understanding, and opportunities associated with MPA designation and implementation (Camargo et al. 2009, Ban et al. 2011). However, when appropriately protected, the regional value of the improved ecological functionality of the protected ecosystem can have far reaching positive impacts ranging from fish spawning sanctuaries increasing regional fish populations (Babcock et al. 2010) to ecotourism providing a new economic viability feature to local economies (Brondo and Bown 2011). However, this is not without risk to the living resources, as evidenced by novice snorkelers negatively impacting seagrass beds in a Mexican MPA (Herrera–Silveira et al. 2010).

MPAs are now established in tropical regions worldwide, but the MPA designation is frequently tied to coral reefs (Gaines et al. 2010, Ban et al. 2011). As these communities are the most prone to debilitating anthropogenic threats, especially climate change, they necessarily need the elevated designation and protection. However, it is the combined and integrative nature of coral reefs, mangroves, and seagrass beds which often provide the regional recovery and resilience of fish and invertebrate populations (Aguilar–Perera and Appeldoorn 2008), but resilience may differ between Caribbean and Pacific reef systems based on functional redundancy and species richness (Roff and Mumby 2012).

Since Caribbean communities seem to lack this system redundancy, the potential for negative human impacts to living resources increases as development and encroachment increase. In St. John, USVI, an increase in housing and commercial development has led to increased land erosion (Macdonald et al. 1997) and a dramatic shift in sediment inputs from natural processes to ones dominated by human activity (Brooks et al. 2007). These external pressures, coupled with natural disturbances and diseases, have led to the degradation of marine communities and negatively impacted fish populations (Rogers and Beets 2001) and coral reefs (Rogers and Miller 2006). Similar trends in coral reef degradation were shown for Honduras (Garcia–Salgado et al. 2008) and may be related to increased erosion and industrial activity (Harborne et al. 2001, Prouty et al. 2008), but limited data in either system exist regarding the potential impacts in affiliated seagrass communities.
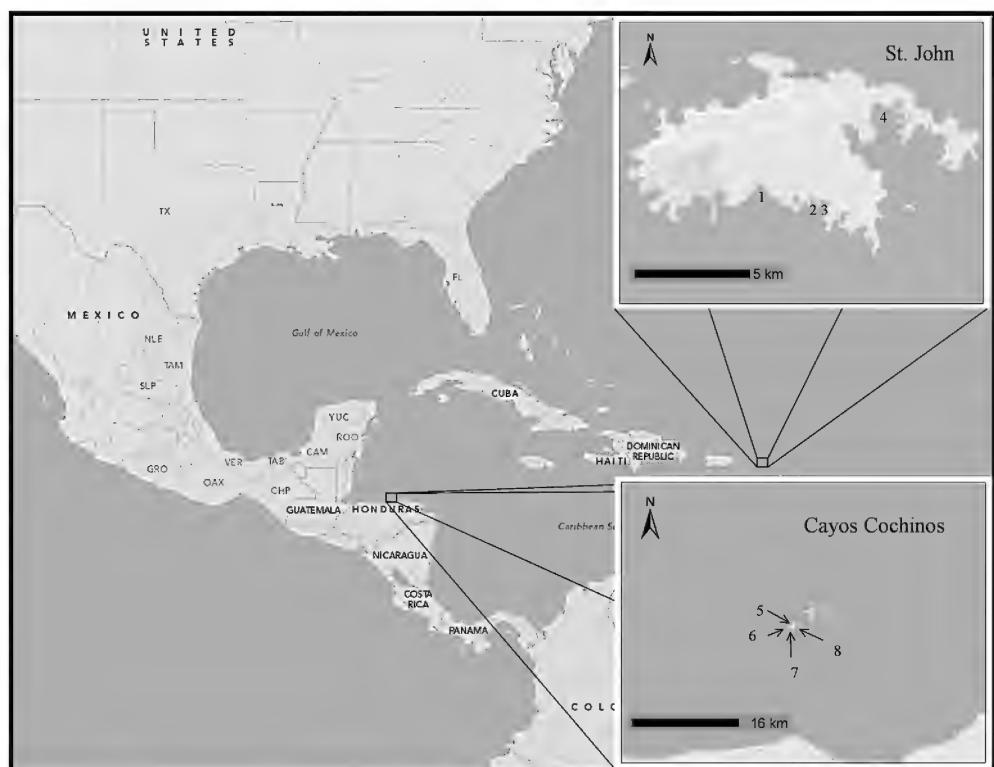

Figure 1. Location of the study sites in St. John, USVI and Cayos Cochinos, Honduras. Site designations are as follows: 1, Reef Bay (REEF); 2, Great Lameshur (GL); 3, Little Lameshur (LL); 4, Hurricane Hole (HH); 5, Jena's Cove (JC); 6, Menor West (MW); 7, Menor South (MS); 8, Menor East (ME).

Tropical seagrass systems represent a complex community composed of a few vascular plant species with numerous algal and marginal coral species. The shared role of primary production and habitat structure provide important resources to fish and invertebrates and are directly linked to elevated fish populations on the adjacent coral reefs (Dorenbosch et al. 2004). Therefore, determining the basic floral community structure is essential to addressing and assessing their role in these integrated communities. However, one of the challenges to determining this structure is the generalized legal nature of MPAs, which restrictively or explicitly prohibit destructive sampling. We present a comparative, non–destructive assessment of seagrass floral communities in Caribbean MPAs in St. Johns. USVI and Hondoras.

MATERIALS AND METHODS

Study sites

Seagrass beds were sampled in two MPAs in the Caribbean. Data were collected in 2009 from the Cayos Cochinos Marine National Monument in Honduras and in 2010 from the Virgin Islands National Park and Coral Reef Monument in St. John, USVI (Figure 1). Each of these sites has substantial governmental protection and restrictions on fishing and other human activities. St. John is an UNESCO designated Biosphere Reserve, while Cayos Cochinos is a Government protected MPA managed by the Honduran Coral Reef Foundation.

Floral community structure was assessed at 4 shallow water seagrass beds in each MPA. Honduran sites surrounding Cayos Menor include Jena's Cove (15°57'30"N, 86°30'14"W), Menor South (15°57'17"N, 86°30'24"W), Menor West (15°57'28"N, 86°30'26"W), and Menor East (15°57'10"N, 86°29'52"W). St. John, USVI sites included Hurricane Hole (18°21'11"N, 64°42'13"W), Great Lameshur (18°19'07"N, 64°43'23"W), Little Lameshur (18°19'11"N, 64°43'40"W), and Reef Bay (18°19'22"N, 64°44'47"W).

Floral assessment

To conduct non–destructive research, a quadrat methodology was employed to determine seagrass demographics and algal species abundance, richness, and diversity. In Honduras, a 50 cm x 50 cm (0.25 m^2) quadrat was used in shallow (<2 m depth), sub–tidal seagrass dominated communities. All algal species were identified to lowest practical taxonomy in the field, with a photographic reference for unidentifiable species during the dive. Algal species were then enumerated in the total quadrat, while seagrass species presence and coverage was assessed visually within the sample quadrat. This technique did not allow for an exact quantification of spatial coverage, but provided a relative dominance assessment among the seagrass species (e.g., monoculture or mixed species beds). A small rectangular sub–grid (8 cm x 9 cm, 0.072m^2) was used to determine shoot density for seagrass species. Research in Honduras was supervised by the Operation Wallacea Dive Program. Dive regulations permit a maximum of 50 minutes for dives, so each site was sampled haphazardly for this time period and as many quadrat samples were collected as possible. Specifically, 14 samples were collected from Menor West (MW), 17 from Menor South (MS), 19 from Jena's Cove (JC), and 18 from Menor East (ME). For research conducted in St. John, a collapsible, portable quadrat was manufactured so that all future sampling protocols would be standardized. To assess the floral characteristics we used 0.09 m^2 quadrats (30 cm x 30 cm) divided into 9 grids of 10 cm x 10 cm. Percent cover of seagrass species was assessed by using a presence–absence ratio of occurrence in the 9 grids. Algal species were identified and counted in the entire quadrat to get a density estimate of each as they comprised part of the community but are not dominant coverage species. The central grid was then investigated and the shoot abundance of each potential seagrass species was counted to calculate a shoot density for each of the quadrats. Total number of samples collected at each site varied in St. John based on logistical considerations. Specifically, 22 samples were collected from Great Lameshur (GL), 25 from Little Lameshur (LL), and 20 from both Reef Bay (REEF) and Hurricane Hole (HH).

Statistical Analysis

While the quadrat sampling methodology differed slightly between Honduras and the St. John sites, the data collected are comparable as coverage and density information were standardized based on the sample quadrat used. Data were standardized for both sites on a per m^2 basis and analyzed using 2–Way Nested ANOVA (SAS®) with site as the independent variable and locations within sites nested in the analysis. Dependant variables related to the spatial coverage and shoot density of turtle grass (*Thalassia testudinum* Banks ex König), manatee grass (*Syringodium filiforme* Kütz), shoal grass (*Halodule wrightii* Ascherson) and density of the abundant algal groups. To assess significant differences among locations within each MPA, we used an LSMeans procedure for discrimination. In all cases, degrees of freedom are $F_{1,147}$ unless otherwise noted. Additionally, to assess floral community structure, we conducted a Similarity of Percentages (SIMPER), an Analysis of Similarities (ANOSIM), and used the non-metric multi-dimensional scaling (MDS) for discrimination outlined in the software program Primer®. Data were 4th root transformed to minimize the bias associated with the numerically dominant seagrass species and abundant algal taxa present in samples. We conducted 2 different analyses, one directed at assessing the total floral community (seagrass and algal taxa) and a second analysis only assessing the community differences associated with algal taxa. The reason the second analysis was carried out was to discriminate the algal community differences among sites from the vascular plant community which dominated the initial analyses.

Results

Characterization of shallow water floral communities showed that both MPAs were dominated by *T. testudinum* with lower quantities of *S. filiforme* (Table 1). Numerous significant differences were seen in seagrass demographics between systems including significantly greater *T. testudinum* spatial coverage (F = 6.04, p < 0.02) and shoot density ($F_{1,146}$ = 123.2, p < 0.0001; df reduced because one shoot count was smudged on a dive slate) in Honduras, while *S. filiforme* spatial coverage (F = 71.5, p < 0.0001) and *H. wrightii* shoot density (F = 7.4, p < 0.008) were significantly greater in St. John. Differences in the density of several algal taxa were also present with significantly higher means for *Dictyota* spp. (F = 6.6, p < 0.02), *Halimeda* spp. (F = 126.2, p < 0.0001), *Valonia* spp. (F = 4.3, p < 0.04), *Penicillus* spp. (F = 12.1, p < 0.001), *Udotea* spp. (F = 8.2, p < 0.005), *Dictyosphaeria cavernosa* (Forsskäl) Børgesen (F = 16.0, p < 0.0001), and *Galaxaura* spp. (F = 8.2, p < 0.005) in Honduras. In contrast, *Padina* spp. (F = 6.7, p < 0.01), *Avrainvillea* spp. (F = 10.8, p < 0.002), and *Wrangelia argus* (Montagne) (F = 4.3, p < 0.04) were significantly greater in St. John. The corals *Porites* spp. (F = 7.5, p < 0.007) and *Montastrea* spp. (F = 4.7, p < 0.03) were significantly greater in Honduras, as they were not found in samples from St. John.

Taxonomic richness did not differ between MPAs (Table 1), suggesting that despite the size differences in the quad-

TABLE 1. Summary of floral and coral demographics for each of the 8 sites investigated. Data represent mean values (± se) of the percent cover of seagrass species and the density of the floral and coral species (#/m²). Asterisks following the taxonomic category indicate significant differences between MPAs (* 0.05, ** 0.01, *** 0.001), while letters following the means indicate significant differences among sites within the MPAs (alpha = 0.05). Site designations are as follows: Reef Bay (REEF); Great Lameshur (GL); Little Lameshur (LL); Hurricane Hole (HH); Jena's Cove (JC); Menor West (MW); Menor South (MS); Menor East (ME).

MPA Region Site	Honduras				St. John			
	MW	MS	JC	ME	HH	GL	LL	REEF
Seagrass taxa								
Thalassia testudinum percent cover*	95.7±2.3	81.8±6.1	86.8±5.1	99.4±0.56	99.4±0.6[a]	68.2±8.8[b]	60.4±9.5[b]	97.2±1.6[a]
T. testudinum shoot density***	873.0±53.3[a]	653.6±70.2[b]	577.5±45.4[b]	678.1±35.5[b]	515.0±50.4[a]	231.8±45.3[c]	184.0±38.6[c]	385.0±27.4[b]
Syringodium filiforme percent cover***	84.3±4.3[a]	55.9±10.8[b]	0[c]	0[c]	95.6±3.9[a]	49.5±9.7[b]	99.1±0.9[a]	35.0±10.1[b]
S. filiforme shoot density	763.9±108.7[a]	392.2±97.6[b]	0[c]	0[c]	315.0±40.6[b]	209.1±72.6[b]	616.0±47.8[a]	160.0±52.0[b]
Halodule wrightii shoot density**	0	0	0	0	0[b]	6.8±4.8[b]	21.6±7.8[a]	0[b]
Algal taxa								
Halimeda***	228.2±88.1[c]	277.8±70.4[bc]	394.7±77.5[b]	709.9±89.0[a]	0.5±0.5	4.1±2.6	19.2±5.1	15.0±5.4
Penicillus***	248.0±73.1[b]	776.1±317.8[a]	87.7±41.4[b]	30.8±14.0[b]	98.0±17.4	7.7±4.2	6.4±3.0	0
Dictyosphaeria cavernosa***	9.9±9.9[b]	392.2±189.5[a]	307.0±91.1[a]	15.4±10.6[b]	1.0±0.7[b]	0[b]	0[b]	5.0±2.5[a]
Dictyota*	0[b]	57.2±43.0[a,b]	102.3±46.1[a]	23.1±12.6[b]	0	11.4±4.4	10.4±3.2	17.5±6.4
Galaxaura**	0[c]	57.2±39.6[b]	73.1±37.5[a]	7.7±7.7[b,c]	0	0	0	0
Caulerpa	0[b]	81.7±57.2[a]	14.6±14.6[b]	0[b]	3.0±2.1	0	0	2.0±1.4
Udotea**	29.8±15.8[a,b]	40.8±25.9[a]	0[b]	30.9±23.9[a,b]	0	0.5±0.5	0	0
Valonia*	9.9±9.9	40.8±40.8	43.8±30.1	0	0	0	0	0
Avrainvillea**	0	0	0	0	2.5±1.4[b]	2.3±1.1[b]	6.4±2.4[a]	1.5±1.5[b]
Padina**	0	0	0	0	0[b]	2.3±1.6[b]	7.6±3.1[a]	0.5±0.5[b]
Wrangelia argus*	0	0	0	0	0[b]	7.3±3.7[a]	0[b]	5.5±4.1[a,b]
Coral								
Porites**	19.8±19.8[a,b]	73.5±58.4[a]	65.8±26.8[a]	0[b]	0	0	0	0
Montastrea*	0	16.3±16.3	7.3±7.3	15.4±10.6	0	0	0	0
Taxa Richness	3.5[a]	3.9[a]	3.3[a]	2.5[b]	3.35[a,b]	2.4[c]	3.64[a]	2.7[b,c]
Floral Diversity***	1.05[a]	1.11[a]	1.02[a]	0.77[b]	0.84[a]	0.41[c]	0.68[a,b]	0.59[b,c]

rats used it had no impact on species richness. However, significant differences were observed among sites within MPAs ($F_{6,147}$ = 5.9, p < 0.0001). Taxonomic diversity was significantly greater from Honduras ($F_{1,147}$ = 47.9, p < 0.0001), but also differed among sites within MPAs (F = 5.5, p < 0.0001). Within MPAs, significant differences among sites were seen for *T. testudinum* shoot density, *S. filiforme* shoot density and spatial coverage, and density of *Dictyota*, *Halimeda*, *Caulerpa*, *Penicillus*, *Udotea*, *Galaxaura*, and *Porites* in Honduras. *Thalassia testudinum* shoot density and spatial coverage, *S. filiforme* shoot density and spatial coverage, *H. wrightii* shoot density, *Padina*, *Dictyosphaeria cavernosa*, *Avrainvillea*, and *Wrangelia argus* density differed among sites in St. John (Table 1).

When floral communities were assessed using ANOSIM (Global R = 0.387, p < 0.001), significant differences existed for all site combinations except Honduras MW and MS (Table 2, upper right). Since all sites were numerically dominated by two seagrass species, discrimination of site differences associated with the algal species was difficult to interpret and the SIMPER analysis documented the dominance of *T. testudinum*, which contributed > 50% to the site similarity (Table 3). A second ANOSIM addressed this by assessing only the algal species present (Global R = 0.373, p < 0.001) and significant differences were seen among almost all sites (Table 2, lower left). When the floral communities were plotted in the MDS, distinct differences can be visualized between Honduras and St. John (Figure 2), but substantial overlap is present due to the dominance of *T. testudinum* with discrimination due to the relative amount of *Halimeda*, *S. filiforme*, and *Penicillus* (Table 3).

TABLE 2. Results of the ANOSIM comparing floral community structure among all locations investigated. Top right section of the table represents the combined seagrass and algal floral community results (R statistic, p value: * = 0.05, ** = 0.01, *** = 0.001), while the lower left section of the table shows the results of the algal only taxa community assessment. Site designations are as follows: Reef Bay (REEF); Great Lameshur (GL); Little Lameshur (LL); Hurricane Hole (HH); Jena's Cove (JC); Menor West (MW); Menor South (MS); Menor East (ME).

	St. John GL	St. John LL	St. John REEF	St. John HH	Honduras MW	Honduras ME	Honduras MS	Honduras JC
St. John GL		0.25***	0.07*	0.26***	0.15*	0.35***	0.20***	0.37***
St. John LL	0.06		0.37***	0.40***	0.25***	0.80***	0.43***	0.80***
St. John REEF	0.1*	0.1		0.51***	0.41***	0.44***	0.37***	0.40***
St. John HH	0.48***	0.54***	0.86***		0.44***	0.87***	0.47***	0.84***
Honduras MW	0.22**	0.29**	0.55***	0.58***		0.70***	0.06	0.66***
Honduras ME	0.44***	0.34***	0.45***	0.80***	0.36**		0.29***	0.09*
Honduras MS	0.31***	0.26***	0.37***	0.64***	0.04	0.18***		0.25***
Honduras JC	0.42***	0.30***	0.30***	0.84***	0.26**	0.09*	0.08*	

DISCUSSION

Global seagrass loss necessarily means that there is a loss in the productivity and ecosystem services which they provide. One of the greatest challenges we face is that while seagrass systems are well studied in many tropical and temperate regions, just as many, if not more, exist in locations where limited data exist. Our findings build upon initial work carried out by Bologna et al. (2008) in St. John and Ogden (1998) and Michot et al. (2002) from Honduras. While Bologna et al. (2008) focused on seagrass community structure, Ogden (1998) focused on the algal communities present on shallow reef sites during the designation of the Cayos Cochinos MPA and Michot et al. (2002) assessed the response and impacts of Hurricane Mitch. Results from our current work demonstrate high spatial coverage of seagrasses in both systems, but substantial differences between MPAs and among sites within the MPAs in regards to both seagrass and algal communities. In particular, *T. testudinum* shoot density and spatial coverage were significantly greater from Honduras compared to St. John, while *S. filiforme* and *H. wrightii* were greater from St. John (Table 1). Additionally, Honduran sites had greater species richness, diversity, and seven algal species with significantly greater densities, while St. John had only four species showing significantly greater densities. As a consequence, the overall floral community structure between sites was significantly different, but the initial SIMPER analysis demonstrated that both Honduras and

TABLE 3. Contributing taxa defining and discriminating the MPAs floral composition based upon SIMPER analysis. Values represent the individual percent contribution to defining the flora responsible for the relationship. Top panel results are based on combined seagrass and algal taxa present in samples, while the lower panel analyses are based only upon the algal taxa present in samples.

| All Flora	St. John		Honduras	
St. John	Average St. John Similarity	49.2%	Average Dissimilarity	58.3%
		Individual Contribution		Individual Contribution
	Thalassia testudinum	51.9%	Halimeda	22.6%
	Syringodium filiforme	34.7%	Syringodium filiforme	19.3%
	Penicillus	4.9%	Thalassia testudinum	13.0%
			Penicillus	12.9%
			Dictyosphaeria cavernosa	8.1%
			Dictyota	7.6%
			Udotea	2.6%
			Galaxaura	2.5%
			Avrainvillea	2.4%
Honduras			Average Honduras Similarity	55.9%
				Individual Contribution
			Thalassia testudinum	54.5%
			Halimeda	28.9%
			Penicillus	6.4%
			Syringodium filiforme	5.9%
Algae Only				
St. John	Average St. John Similarity	24.3%		
		Individual Contribution		
	Penicillus	44.9%		
	Dictyota	22.1%		
	Halimeda	19.2%		
	Avrainvillea	7.0%		
Honduras	Average Dissimilarity	79.85%	Average Honduras Similarity	43.98%
		Individual Contribution		Individual Contribution
	Halimeda	34.5%	Halimeda	74.0%
	Penicillus	20.5%	Penicillus	17.3%
	Dictyota	11.5%		
	Dictyosphaeria cavernosa	10.6%		
	Avrainvillea	4.3%		
	Udotea	4.1%		
	Galaxaura	3.2%		
	Padina	3.1%		
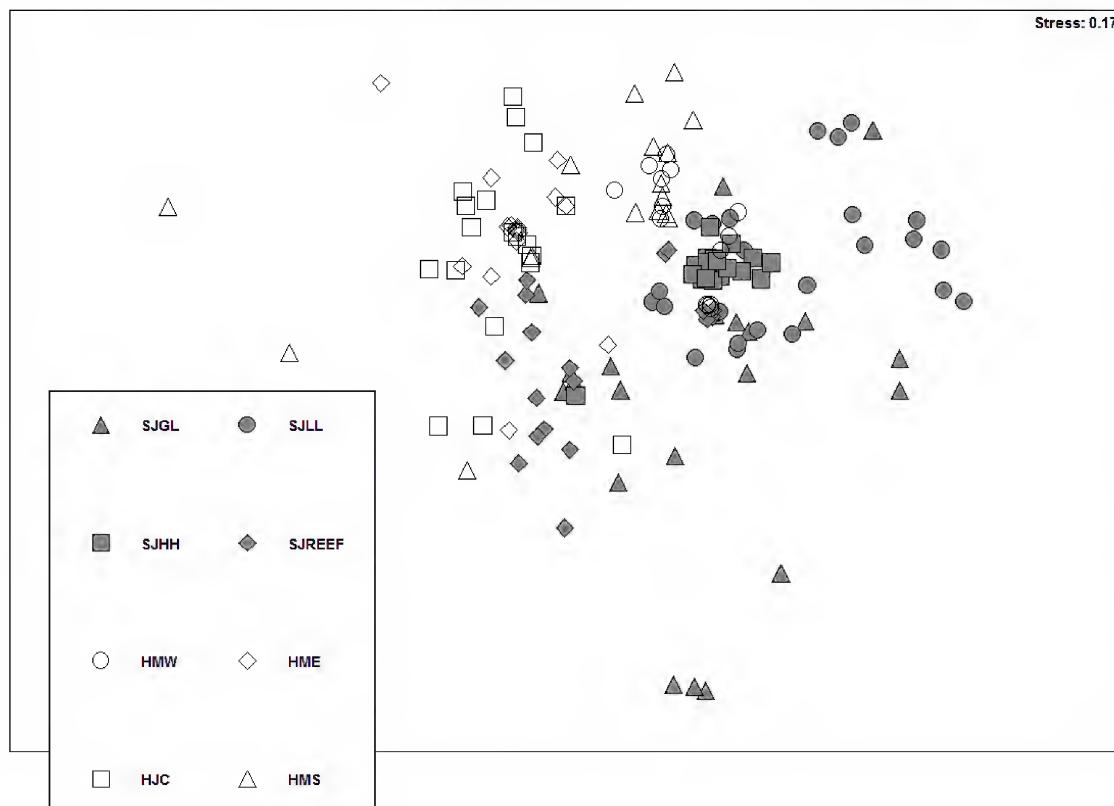
Figure 2. MDS plot of floral community structure between Honduras and St. John. Open symbols represent samples from Honduras, filled symbols represent samples from St. John. Specific sites designated with prefixes of H for Honduras and SJ for St. John in front of location abbreviations. Site designations are as follows: Reef Bay (REEF); Great Lameshur (GL); Little Lameshur (LL); Hurricane Hole (HH); Jena's Cove (JC); Menor West (MW); Menor South (MS); Menor East (ME).

St. John communities were dominated by *T. testudinum*. In general, our results concur with the previous studies demonstrating dominance by *T. testudinum* with lesser amounts of *S. filiforme*, while *Halimeda* and *Penicillus* dominate the algal groups. Michot et al. (2002) documented *H. wrightii* growth in shallow disturbed regions of Honduras, but it was absent from our current assessment of seagrass community structure. When the seagrass species were eliminated from the analyses, the pattern was dominated by *Penicillus* in St. John and *Halimeda* in Honduras. Consequently, while these systems showed remarkable similarities in species presence, they maintained unique characteristics with an average dissimilarity of 79.8%. Beyond algal differences, the presence of small coral colonies present in Honduras suggests substantially different shallow benthic community structure.

The *Porites* and *Montastrea* colonies present in Honduras suggest healthier reef structure, which corresponds to the differences in MPAs. St. John is predominately a United States National Park with designated protected regions associated with the coastal marine systems, but many residents and visitors inhabit the island. This leads to substantially greater visitation and infrastructure. In fact, the development on St. John has lead to considerable erosion (Brooks et al. 2007), potentially impacting both reef and seagrass communities. Additionally, much of the island is not on a public sewage system, which creates the potential for nutrient loading in the system, and large desalinization plants exist associated with the tourist developments to serve the high water de-

mands. The Honduras MPA has extremely limited development and infrastructure with only limited access to tourists. As such, the anthropogenic pressures are considerably less on the adjacent marine communities, but coastal development in Honduras is progressing (Harborne et al. 2001) and stressors (e.g., sediments, heavy metals) from terrestrial and industrial sources will continue (Prouty et al. 2008). Supporting evidence for greater reef degradation on St. John comes from Rogers and Miller (2006) who demonstrated low coral spatial coverage (10–12%) on reefs in that region compared to the relatively higher coverage seen in Honduras (10–29%, García–Salgado et al. 2008, Bologna et al. 2012). Rogers and Miller (2006) suggest that the coral decline was most likely due to natural and human induced impacts such as hurricanes, over–fishing, and continued development. While these stressors impact Honduran reefs, the reduced anthropogenic stress may play a substantial role in the resilience of the Cayos Cochinos reefs (Carilli et al. 2009), and therefore provide greater recruitment potential for corals or greater environmental stability for recruitment, survival, and growth of *Porites* and *Montastrea*. However, Green et al. (2008) suggest elevated *Porites astreoides* (Lamarck) abundance may be due to the declining reefs creating open space for this species. This argument is not plausible for *Porites* found among the dense grass beds and the greater health of the Honduran reefs provides a proximal mechanism for greater potential recruitment and consequent presence in these grass beds.

The differences observed between MPAs is not unex-

pected, however the differences in flora communities seen among sites within each MPA was. For St. John, 3 of the 4 sites are relatively exposed, but HH is quite protected and had significantly greater *T. testudinum* shoot density compared to the other sites, as well as an order of magnitude greater abundance of *Penicillus* spp., but little *Halimeda* and no *Dictyota* or *Padina*. Site LL was distinguished by its low abundance of *T. testudinum* and significantly greater *S. filiforme*, *H. wrightii*, *Padina*, and *Avrainvillea*. The last 2 sites are principally distinguished by their relative abundances of *T. testudinum* and *S. filiforme*. Why these major differences exist among sites might be explained to a limited degree by relative oceanic exposure, but our data from adjacent bays showing such defined flora communities needs further investigation. Bologna et al. (2008) showed differences in flora communities and organic carbon among GL, LL, and HH, but REEF was not sampled, nor was sediment size structure and composition analyzed. In Honduras, some of the within–MPA differences were seen in the relative cover and shoot density of *T. testudinum*, but a major distinguishing factor was that only half the sites had *S. filiforme* present. Sites with *S. filiforme* also had high densities of *Penicillus*, but lower densities of *Halimeda*. Collectively, these differences are evident

at the community level (Figure 2), but additional research is needed to address community structure and the relatively low species richness observed in both MPAs.

One broad challenge in assessing communities within MPAs relates to the ability to collect quantitatively defensible data. The use of quadrats provides essential information regarding the plant density, species richness, and floral diversity. However, biomass, shoot structure, presence of rare algal taxa and other quantitative data are only obtainable through destructive methods, frequently prohibited in many MPAs. This work provides essential information necessary to describe a baseline of seagrass community structure, while we develop long–term strategies for the monitoring of these habitats from these and other MPAs. This type of assessment is currently being completed through Seagrass.net, but different protocols may be needed to cover greater areas to develop robust region–wide data sets from areas which do not have established long–term monitoring plans in place or do not have the organized local communities trained to carry out consistent and long–term monitoring. We recognize the limitations of non–destructive sampling, but affirm the efficacy of using this method to gather quantifiable and comparable data sets among numerous sites.

LITERATURE CITED

Aguilar–Perera, A. and R.S. Appeldoorn. 2007. Spatial distribution of marine fishes along a cross–shelf gradient containing a continuum of mangrove–seagrass–coral reefs off southwestern Puerto Rico. Estuarine Coastal and Shelf Science 76:378–394.

Babcock, R., N. Shears, A. Alcala, N. Barrett, G. Edgar, K. Lafferty, T. McClanahan, and G. Russ. 2010. Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. Proceedings of the National Academy of Science 107:18256–18261.

Ban, N.C., V.M. Adams, G.R. Almany, S. Ban, J.E. Cinner, L.J. McCook, M. Mills, R.L. Pressey, and A. White. 2011. Designing, implementing and managing marine protected areas:

emerging trends and opportunities for coral reef nations. Journal of Experimental Marine Biology and Ecology 408:21–31

Bologna, P., R. Papagian, S. Regetz, and C. Dale. 2008. Assessment of turtle grass (*Thalassia testudinum* ex Banks Konig) community structure in a UNESCO Biosphere Reserve. Journal of Experimental Marine Biology and Ecology 365:148–155.

Bologna, P., L. Webb–Wilson, P. Connelly, and J. Saunders. 2012. A new baseline for *Diadema antillarum*, *Echinometra viridis*, *E. lucunter*, and *Eucidaris tribuloides* populations within the Cayos Cochinos MPA, Honduras. Gulf and Caribbean Research 24:1–5.

Bosire, J.O., F. Dahdouh–Guebas, M. Walton, B.I. Crona, R.R. Lewis, C. Field, J.G. Kairo, and N. Koedam. 2008. Functionality of restored mangroves: a review. Aquatic Botany 89:251–259.

Brondo, K. and N. Bown. 2011. Neoliberal conservation, Garifuna territorial rights and resource management in the Cayos Cochinos Marine Protected Area. Conservation and Society 9:91–105.

Brooks, G., B. Devine, R. Larson, and B. Rood. 2007. Sedimentary development of Coral Bay, St. John, USVI: a shift from natural to anthropogenic influences. Caribbean Journal of Science 43:226–243.

Camargo, C., J. Maldonado, E. Alvarado, R. Moreno–Sanchez, S. Mendoza, N. Manrique, A. Mogollon, J. Osoria, A. Grajales, and J. Sanchez. 2009. Community involvement in management for maintaining coral reef resilience and biodiversity in southern Caribbean marine protected areas. Biodiversity and Conservation 18:935–956.

Carilli, J.E., R.D. Norris, B.A. Black, S.M. Walsh, and M. McField. 2009. Local stressors reduce coral resilience to Bleaching. PLoS ONE 4:e6324. doi:10.1371/journal.pone.0006324.

Dorenbosch, M., M.C. van Riel, I. Nagelkerken, and G. van der Velde. 2004. The relationship of reef fish densities to the proximity of mangrove and seagrass nurseries. Estuarine Coastal and Shelf Science 60:37–48.

Gaines, S.D., C. White, M.H. Carr, and S.R. Palumbi. 2010. Designing marine reserve networks for both conservation and fisheries management. Proceedings of the National Academy of Science 107:18286–18293.

García–Salgado, M.A., G.G. Nava–Martinez, M. Vazquez, N.D. Jacobs, I. Majil, A. Molina–Ramirez, B. Yañez–Rivera, A. Cubas, J.J. Dominquez–Calderon, W. Hadaad, M.A Maldonado, and O. Torres. 2008. Declining trend on the Mesoamerican Reef System Marine Protected Areas. Proceedings of the 11th International Coral Reef Symposium, Ft. Lauderdale, FL, USA, 7–11 July 2008, p. 883-888.

Green, D., P. Edmunds, and R. Carpenter. 2008. Increasing relative abundance of *Porites astreoides* on Caribbean reefs mediated by an overall decline in coral cover. Marine Ecology Progress Series 359:1–10.

Harborne, A., D. Afzal, and M. Andrews. 2001. Honduras: Caribbean coast. Marine Pollution Bulletin 42:1221–1235.

Herrera–Silveira, J.A., J. Cebrian, J. Hauxwell, J. Ramirez–Ramirez, and P. Ralph. 2010. Evidence of negative impacts of ecological tourism on turtlegrass (*Thalassia testudinum*) beds in a marine protected area of the Mexican Caribbean. Aquatic Ecology 44:23–31.

MacDonald, L., D. Anderson, and W. Dietrich. 1997. Paradise threatened: land use and erosion on St. John US Virgin Islands. Environmental Management 21:851–863.

Michot, T.C., J.N. Burch, A. Arrivillaga, P.S. Rafferty, T.W. Doyle, and R.S. Kemmerer. 2002. Impacts of Hurricane Mitch on seagrass beds and associated shallow reef communities along the Caribbean coast of Honduras and Guatemala: USGS Open File Report 03–181, National Wetlands Research Center, Lafayette, LA, USA, 65 p.

Mumby, P., A. Hastings, and H. Edwards. 2007. Thresholds and the resilience of Caribbean coral reefs. Nature 450:98–101.

Ogden, N., 1998. Checklist of marine benthic algae in the Cayos Cochinos archipelago, Honduras. Revista de Biología Tropicál 46 (suppl. 4):81–87.

Orth R.J., T. Carruthers, W.C. Dennison, C.M. Duarte, J.W. Fourqurean, K.L., Heck, A.R. Hughes, G.A. Kendrick, W.J. Kenworthy, S. Olyarnik, F.T. Short, M. Waycott, and S. Williams. 2006. A Global Crisis for Seagrass Ecosystems. BioScience 56:987-996.

Pandolfi, J.M., R. Bradbury, E. Sala, T. Hughes, K. Bjorndal, R. Cooke, D. McArdle, L. McClenachan, M. Newman, G. Paredes, R. Warner, and J. Jackson. 2003. Global trajectories of the long–term decline of coral reef ecosystems. Science 301:955–958.

Prouty, N., K. Hughen, and J. Carilli. 2008. Geochemical signatures of land–based activities in Caribbean coral surface samples. Coral Reefs 27:727–742.

Rivera–Monroy, V.H., R. Twilley, D. Bone, D. Childers, C. Coronado–Molina, I. Feller, J. Herrera–Silveira, R. Jaffe, E. Mancera, E. Rejmankova, J. Salisbury, and E. Weil. 2004. A conceptual framework to develop long–term ecological research and management objectives in the wider Caribbean region. Bioscience 54:843–856.

Roff, G. and P. Mumby. 2012. Global disparity in the resilience of coral reefs. Trends in Ecology and Evolution 27:404–413.

Rogers, C. and J Beets. 2001. Degradation of marine ecosystems and decline of fishery resources in marine protected areas in the US Virgin Islands. Environmental Conservation 28:312–322.

Rogers, C. and J. Miller. 2006. Permanent 'phase shifts' or reversible declines in coral cover? Lack of recovery of two coral reefs in St. John, US Virgin Islands. Marine Ecology Progress Series 306:103–114.

Short, F., B. Polidoro, S. Livingstone, K. Carpenter, S. Bandeira, J. Bujang, H. Calumpong, T. Carruthers, R. Coles, W. Dennison, P. Erftemeijer, M. Fortes, A.Freeman, T. Jagtap, A. Kamal, G. Kendrick, W. Kenworthy, Y. La Nafie, I. Nasution, R. Orth, A. Prathep, J. Sanciangco, B. van Tussenbroek, S. Vergara, M. Waycott, and J. Zieman. 2011. Extinction risk assessment of the world's seagrass species. Biological Conservation 144:1961–1971.

Thampanya, U., J.E. Vermaat, S. Sinsakul, and N. Panjapitkkul. 2006. Coastal erosion and mangrove progradation of southern Thailand. Estuarine Coastal and Shelf Science 68:75–85.

Unsworth, R. and L. Cullen. 2010. Recognizing the necessity of Indo–Pacific seagrass conservation. Conservation Letters 3:63–73.

Valiela, I., J.L. Bowen, and J.K. York. 2001. Mangrove forests: one of the world's threatened major tropical environments. BioScience 51:807–815.

Waycott, M., C. Duarte, T. Carruthers, R. Orth, W. Dennison, S. Olyarnik, A. Calladine, J. Fourqurean, K. Heck, A.R. Hughes, G. Kendrick, W. Kenworthy, F. Short, and S. Williams. 2009.

Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proceedings of the National Academy of Science 106:12377–12381.

Wilkinson, C. and B. Salvat. 2012. Coastal resource degradation in the tropics: does the tragedy of the commons apply for coral reefs, mangrove forests and seagrass beds. Marine Pollution Bulletin 64:1096–1105.

Spatial and Size Distribution of Red Drum Caught and Released in Tampa Bay, Florida, and Factors Associated with Post-Release Hooking Mortality

Kerry E. Flaherty
Florida Fish and Wildlife Conservation Commission

Brent L. Winner
Florida Fish and Wildlife Conservation Commission

Julie L. Vecchio
Florida Fish and Wildlife Conservation Commission

Theodore S. Switzer
Florida Fish and Wildlife Conservation Commission

SPATIAL AND SIZE DISTRIBUTION OF RED DRUM CAUGHT AND RELEASED IN TAMPA BAY, FLORIDA, AND FACTORS ASSOCIATED WITH POST–RELEASE HOOKING MORTALITY

Kerry E. Flaherty*, Brent L. Winner, Julie L. Vecchio[1], and Theodore S. Switzer

*Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute, 100 8th Avenue Southeast, St. Petersburg, FL 33701 USA; [1]Current address: 444 51st Avenue South, St. Petersburg, FL 33705 USA; *Corresponding author, email: kerry.flaherty@myfwc.com*

ABSTRACT: The recreational fishery for red drum (*Sciaenops ocellatus*) in Florida is unusual in that most red drum targeted are immature and caught within estuarine waters. Current state regulations rely exclusively on bag and size limits, resulting in the release of a large proportion of captured individuals. This study employed hook–and–line sampling conducted monthly in Tampa Bay, Florida and catch–and–release mortality experiments to determine the spatial and size distribution of red drum and the mortality rate of released fish, respectively. Of the 1,405 red drum collected, more than 70% were smaller than the minimum legal size (457 mm standard length (SL)). Size structure of red drum varied spatially and reflected ontogenetic patterns of habitat use. Data collected during catch–and–release mortality experiments were analyzed to identify factors associated with mortality. A total of 251 red drum (203–618 mm SL) were caught and held for 48 h during 9 experiments, with an overall mortality rate of 5.6%. Higher water temperature and anatomical hook position were significantly correlated with mortality; lip–hooked fish had the lowest mortality rate, while throat–hooked fish had the highest. Although hook type was not correlated with mortality, it did influence whether a fish was deep–hooked. Fish caught by J–hooks were more likely to be deep–hooked than those caught by circle hooks. Catch–and–release fishing is an effective management tool for reducing take but may contribute to short–term mortality, especially in warm, subtropical estuaries.

KEY WORDS: catch–and–release, *Sciaenops ocellatus*, J–hooks, circle hooks

INTRODUCTION

The red drum (*Sciaenops ocellatus*) fishery is one of the most popular recreational estuarine fisheries in the southeastern United States and is unusual in that most harvested red drum are immature (Murphy and Crabtree 2001, Switzer et al. 2009). Anglers target red drum throughout the year in estuaries along the south Atlantic and Gulf of Mexico (GOM) coasts (Murphy and Munyandorero 2008). In Florida, the harvest of red drum is managed with restrictive size and bag limits to ensure that adequate numbers of fish survive to maturity and recruit to nearshore spawning populations. Current state regulations include a daily bag limit of one fish per person in southern waters and 2 fish per person along the northwest and northeast coasts, a slot limit of 457–686 mm (18–27 inches) total length (TL), and prohibition of commercial harvest. Red drum ontogeny, especially their estuarine life history stages, combined with restrictive bag and size limits, results in a fishery characterized as predominantly catch–and–release.

Red drum spawn from mid–August through November near bay mouths and inlets and in nearshore continental shelf waters (Yokel 1966, Mercer 1984, Murphy and Taylor 1990). Recruitment of juveniles into nursery areas begins in September and continues through February, with peak recruitment in October and November (Peters and McMichael 1987, Daniel 1988). Oligohaline backwater areas (e.g., tidal creeks and rivers) have been documented as primary nursery habitats for juvenile red drum in Tampa Bay and other estuaries (Peters and McMichael 1987, Wenner 1992, Bacheler et al. 2008). Newly settled red drum in seagrass and salt marsh habitat in other GOM estuaries also demonstrated significantly higher growth rates and abundance than in non–vegetated or oyster substrates (Baltz et al. 1998, Stunz et al. 2002a, b). Red drum grow quickly during their first year, reaching about 342 mm TL (Murphy and Taylor 1990). Peters and McMichael (1987) observed that as young red drum increase in size and age, they gradually move from oligohaline habitats into areas of higher salinity. Between ages 1 and 4, red drum use a wide variety of estuarine habitats, including oyster bars, flooded salt marsh, seagrass flats, and mangrove shorelines (Peters and McMichael 1987, Wenner 1992). By age 5 (~780 mm standard length (SL)), most GOM red drum mature, leave their natal estuary, and move into nearshore coastal waters (Murphy and Taylor 1990, Murphy and Crabtree 2001).

The frequency of catch–and–release fishing for saltwater species, including red drum, has increased substantially in recent decades. By the early 1990s the estimated number of released red drum in Florida had reached levels 4 to 5 times the harvest (Murphy and Munyandorero 2008). According to the National Oceanographic and Atmospheric Administration's (NOAA) Marine Recreational Fisheries Statistics Survey (MRFSS), an estimated 2.3 – 4.5 million red drum were caught each year in Florida waters from 2005 to 2009 (NOAA 2008). Of these, only 12–16% were harvested; the remaining 2 – 4.2 million were released. The size of red drum

being caught by anglers, as well as the survival of released individuals, has significant bearing on the inshore population and a strong influence on the level of escapement into nearshore adult populations.

Several factors may influence post–release mortality of red drum, including variations in environmental characteristics and fishing techniques. Reviews of catch–and–release publications showed that environmental characteristics (e.g. temperature, depth, dissolved oxygen (DO); Muoneke and Childress 1994), anatomical site of hooking (Aguilar et al. 2002, Aalbers et al. 2004), amount of bleeding (Fabrizio et al. 2008), hook size (Cooke et al. 2005), and whether the hook was removed (Muoneke and Childress 1994) all may influence short–term (48-72 h) mortality. Several studies have shown that deep–hooking (i.e., hooking in the fish esophagus or stomach) is more common with J–hooks than with circle hooks (Aalbers et al. 2004, Beckwith and Rand 2005, Vecchio and Wenner 2007) and J–hooks are also more highly correlated with greater short–term mortality than was the use of circle hooks (Taylor et al. 2001, Bartholomew and Bohnsack 2005, Vecchio and Wenner 2007). Removing deeply–embedded hooks may also affect survival by increasing handling time and causing additional tissue damage and bleeding (Taylor et al. 2001, Vecchio 2006).

Although the relationship between catch–and–release fishing for red drum and mortality has been studied in other parts of their range (Aguilar et al. 2002, Matlock et al. 1993, Vecchio and Wenner 2007), no studies have been published from experiments conducted in waters as far south as Florida. The Tampa Bay watershed is one of the most accessible and heavily urbanized estuaries in the state, is home to more than 2 million people, and is visited by millions more each year (United States Geological Survey 2008). Therefore, the potential for high recreational fishing pressure on red drum makes it a relevant study area. Accordingly, the current study aims to (1) determine the size and spatial distribution of red drum available to the recreational fishery within the Tampa Bay estuary, (2) identify factors contributing to short–term post–release hooking mortality, and (3) relate these results to long–term tag return data for red drum released alive.

Materials and Methods
Monthly hook–and–line sampling

Hook–and–line surveys were conducted monthly from April 2005 through December 2007 at 10 stations rang-
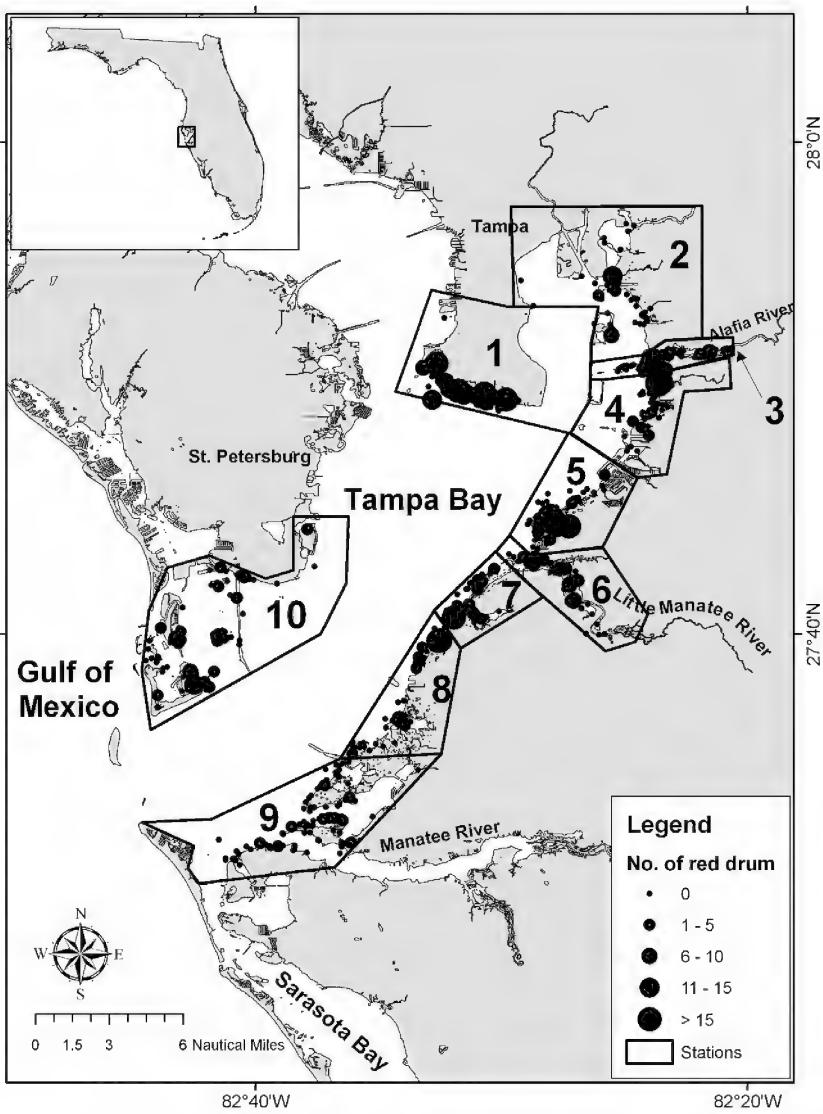

Figure 1. Distribution of monthly hook-and-line sampling sites and number of red drum captured in Tampa Bay, Florida (black circles, April 2005–December 2007). Multiple sites were fished per sampling trip. Sampling stations selected for monthly hook-and-line sampling are labeled 1–10. Catch-and-release mortality experiments were conducted in stations 1, 3, 4, and 10.

ing from the interbay peninsula (Station 1) and the eastern shoreline of Tampa Bay to the northern and southern shorelines of lower Tampa Bay (Figure 1). These surveys were used to document spatial differences in abundance and size structure and thus describe the population of red drum accessible to the recreational fishery. One sampling trip per station was conducted each month throughout the study period. Shallow water habitats within the estuary known to be utilized by red drum, such as oyster bars, seagrass beds, vegetated shorelines, and tidal creeks and rivers were targeted for fishing. The particular habitat targeted for fishing was recorded, and a dominant shore type (overhanging vegetation, emergent vegetation, structure, or other), bottom vegetation type (algae, submerged aquatic vegetation (SAV), structure, or other), and substrate (sand, mud, or structure) were assigned to each sampling site. Each habitat category was assigned as

dominant if it characterized > 50% of the fished area at each sampling site.

Sampling crews consisted of 2 to 3 researchers and volunteer anglers with similar fishing expertise who fished for a minimum of 4 h within their assigned station. Anglers used light rods and tackle to best mimic that used in the red drum recreational fishery and fishing intensity on each trip was comparable. Spinning reels were outfitted with 12 lb test monofilament line and a short monofilament leader; however, each angler was allowed to choose between a 1/0 non–offset circle hook and a 1/0 J–hook. Anglers most frequently used circle hooks and live bait (pink shrimp *Farfantepenaeus duorarum*, scaled sardine *Harengula jaguana*, Atlantic thread herring *Opisthonema oglinum*, or pinfish *Lagodon rhomboides*). However, J–hooks were used regularly, and artificial baits were used occasionally, especially at times of the year when live bait was not readily available. Red drum were measured for SL (mm) and TL (mm), tagged externally with a Hallprint® dart tag inserted between the second and third pterygiophores of the first dorsal fin, and released at the sampling site. Catch data were recorded for all fish and included hook type (J–hook or circle hook), hook position, bait used, whether the hook had been removed, and release condition. Hook positions were defined as follows: "lip" indicated being hooked in the lip or corner of the mouth, "inside mouth" indicated being hooked in the buccal cavity, "throat" indicated being hooked just ahead of pharyngeal teeth, "gut" indicated being hooked beyond pharyngeal teeth, and "other" indicated being hooked in any other position. Release condition was considered "good" if the fish swam away immediately, "fair" if it struggled for several seconds before swimming away, "poor" if it struggled for several minutes before swimming away, or "dead" upon release.

Catch–and–release mortality experiments

To collect a wide size range of red drum and evaluate the potential interaction of season and environmental differences on hooking mortality, 9 replicate catch–and–release mortality experiments were conducted in 3 distinct locations within Tampa Bay from November 2005 to March 2008. Experiments were conducted in shallow seagrass beds and tidal creeks near the interbay peninsula in upper Tampa Bay (Station 1; n = 5), in the Alafia River and associated tidal creeks (Stations 3 and 4 combined; n = 3), and in seagrass beds and mangrove shorelines in lower Tampa Bay (Station 10; n = 1; Figure 1). A station was targeted for sampling if the monthly hook–and–line sampling indicated that sufficient numbers (n > 10) of red drum were recently captured to conduct an experiment that would meet assumptions regarding distribution of errors in statistical analyses. For each experiment, at least 2 crews, each consisting of 2 to 4 scientists, volunteer anglers, and local fishing guides, fished in the designated area. Anglers used either a 1/0 J–hook or 1/0 non–offset circle hook and, unlike the monthly hook–and–line sam-

pling trips, exclusively used live bait to control for potential bait–associated differences in mortality (Muoneke and Childress 1994, Bartholomew and Bohnsack 2005). Effort was adjusted during each experiment to ensure that about the same number of individuals were collected with each hook type. Anglers collected red drum using fishing gear identical to that used during the monthly hook–and–line sampling.

Depending on the number of fish collected during each experiment, one (≤ 40 fish) or 2 (> 40 fish) holding pens were deployed in a centralized location within each study area and served as the main containment locations for test fish. The holding pens were cylindrical (5.5 m in diameter by 2.4 m deep), and constructed of 6.4 mm stretch knotless nylon mesh and could have confined many more than 40 fish without overcrowding. Each pen was secured by 8 galvanized poles inserted into rings around the net pen in an area where water depth was at least 1 m at all tidal stages.

As with monthly hook–and–line sampling, catch data (see list earlier) were recorded for all fish. The length of each fish was measured (SL and TL mm) and additional data were recorded during mortality experiments to document handling of individual fish. Anglers recorded fight time (number of seconds from when a fish was hooked to when it was brought into the boat) and handling time (number of seconds a fish was out of the water for measuring and tagging) for each fish. Every fifth red drum collected by each sampling crew was individually identified by external features or markings and left untagged as a control for estimation of tagging–associated mortality; all other fish were tagged for identification. Fish were held in a live well aboard the fishing vessel for no more than 1 hour before being transferred to the larger holding pen. Water temperature (°C), DO (mg/L), and salinity were recorded at each fishing site and periodically at the holding pens (a minimum of every 12 h).

Red drum were held in the holding pen for at least 48 h, a time period that has been shown to be sufficient for documenting short–term mortality (Bugley and Shepherd 1991, Matlock et al. 1993, Murphy et al. 1995, Taylor et al. 2001). The holding pen was thoroughly checked by a snorkeler each day at dawn and dusk. Any dead red drum were removed and the date and time recorded. All dead fish were returned to the laboratory for further examination and evaluated for possible cause of death. After 48 h, the fish used as tagging controls were tagged for identification, condition was noted for all surviving fish, and all fish were released within the sampling area.

Statistical methods

Total effort and catch data were summarized for red drum collected during monthly hook–and–line sampling and catch–and–release experiments conducted within Tampa Bay. Fishing locations were plotted in a geographic information system to examine the spatial coverage of sampling. Habitat types and water quality characteristics were summa-

TABLE 1. Summary of the number of red drum collected during hook-and-line sampling by station (N) including the mean number captured per trip (mean, standard error (se), maximum, frequency of occurrence (% Freq.)) and the ratio of red drum captured using circle vs. J-hooks (C:J).

Station	N	Number of red drum per trip			C:J
		Mean ± se	Maximum	% Freq.	
1	465	6.2 ± 1.1	58	77.3%	3.5
2	76	2.1 ± 0.6	16	45.9%	2.6
3	140	3.7 ± 1.3	34	65.8%	3.7
4	225	6.4 ± 2.5	70	45.7%	2.3
5	159	4.7 ± 2.2	54	41.2%	4.7
6	68	2.0 ± 0.6	18	52.9%	10.3
7	65	1.9 ± 0.8	24	47.1%	4.9
8	131	3.6 ± 1.1	31	51.4%	3.2
9	15	0.5 ± 0.2	4	27.3%	14.0
10	61	1.6 ± 0.5	14	43.2%	5.1
Total	**1,405**	**3.3 ± 1.1**	**70**	**49.8%**	**3.5**

rized and compared by station using Chi–square tests and multiple factor ANOVAs, respectively. Proportions of red drum collected were summarized by station, hook type, and hook position. Length frequency histograms were plotted by station to further explore the spatial distribution of the sizes of red drum captured during hook–and–line sampling and to describe the size structure of red drum that survived or died during catch–and–release experiments. Differences in mean size among hook–and–line stations and size distributions between hook types with which red drum were captured were tested using Kruskal–Wallis and Kolmogorov–Smirnov non–parametric tests, respectively. The proportions of individuals falling within the current regulated slot size were summarized and displayed as length frequency histograms in SL based on SL/TL length regressions calculated from long–term fisheries–independent monitoring data conducted in Tampa Bay (McMichael 2010).

For catch–and–release mortality experiments, logistic regression was used to determine whether a variety of independent variables were significantly related to the probability of mortality. To assess the probability of mortality, categorical variables included whether the fish was an untagged control, the hook type, the hook position, bait type (fish or shrimp), the fish condition, and whether the hook had been removed. Covariates included mean water temperature, salinity, DO, and water depth assessed at the holding pen during each experiment, fight time, handling time, transport time, and fish length. In addition to these main effects, the potentially confounding interaction effects of hook position x hook type and water temperature x dissolved oxygen were also examined. The probability of mortality (M) was determined with the following equation: $P(M) = e^u / (1 + e^u)$; where e = base of the natural logarithm and u = linear function of the independent variables (Sokal and Rohlf 1995). A forward selection method was used to add variables to the model that

were significant at $p \leq 0.05$. Once a significant variable was entered in the model, it was not removed from the model. The process was repeated until none of the remaining variables met the specified level for entry. The probability of deep–hooking (throat– or gut–hooked) was also determined with similar methodology since previous studies have associated a higher incidence of deep–hooking when anglers use J–hooks (Aalbers et al. 2004, Beckwith and Rand 2005, Vecchio and Wenner 2007). To assess the probability of deep–hooking, hook type and bait type were treated as categorical variables, and the covariates included mean water temperature, salinity, DO, and depth, as well as fight time and fish length.

Long–term survival was estimated from tag returns reported from the beginning of the study period to October 2012 for red drum released after mortality experiments. A relative risk analysis was used to compute relative survival (S) using a technique described by Hueter et al. (2006): $S = R_e / R_u$; where R_e and R_u are the recapture rates for red drum that were either exposed (e) or unexposed (u) to a hook position or hook type that may reduce the possibility of survival. Recapture rates for each exposure group were calculated as the number of red drum released alive after mortality experiments that were recaptured, divided by the number not recaptured. This calculation was based on the assumption that after the initial 48 h holding period lip–hooked fish and those captured by circle hooks had a 100% survival rate and were therefore "unexposed." Mantel–Haenszel 95% confidence intervals for relative survival were calculated (Hueter et al. 2006). All statistics were calculated using SAS version 9.1.3 (SAS Institute Inc. 2006) and were considered significant if $p \leq 0.05$.

RESULTS

Monthly hook–and–line sampling

Three hundred and twenty–four hook–and–line sampling trips were conducted between April 2005 and December 2007 (Figure 1). A total of 1,405 red drum were caught during these trips (Table 1). Red drum were caught on about half of all fishing trips and in all designated fishing stations throughout Tampa Bay (Table 1, Figure 2). Habitat types and water quality characteristics, except temperature (ANOVA, F = 0.86, p = 0.571), were significantly different among stations (X^2 and ANOVA tests p < 0.05). The habitat most targeted for fishing was characterized by overhanging vegetation (predominantly mangroves) that either contained SAV or was unvegetated with substrates of mud or sand (Table 2). By fishing trip, the highest mean number of red drum were caught in Stations 1 and 4 (6.2 and 6.4 red drum per trip, respectively; Table 1). The fewest red drum per trip (0.5)
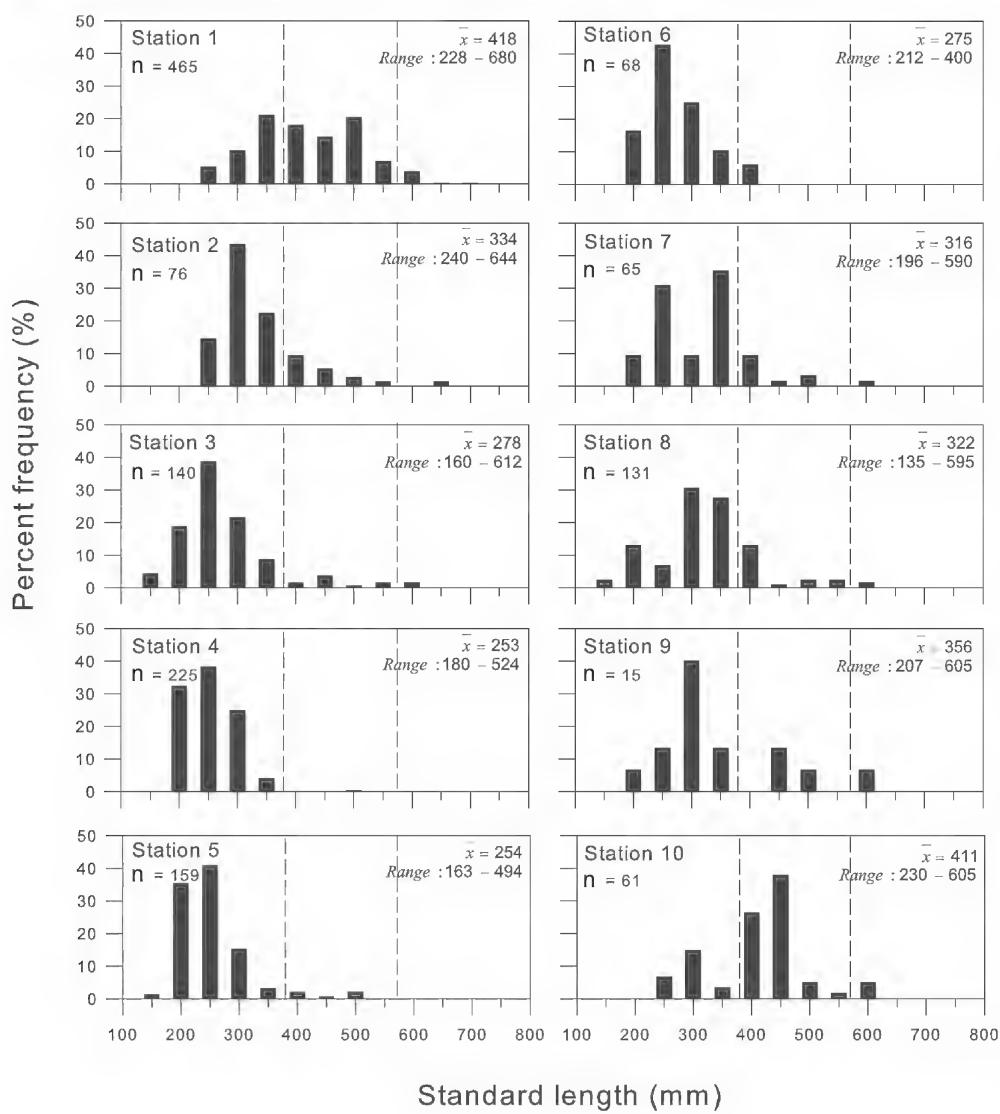
Figure 2. Length frequency (by 50 mm size bins) of red drum caught in Tampa Bay with hook-and-line gear, by sampling station (April 2005 – December 2007; includes fish used for mortality experiments). The Florida legal slot size, estimated from SL/TL length regressions, is indicated by vertical dashed lines on each plot. n = number of red drum collected within that station, $\bar{x}$ = mean SL, range = minimum – maximum SL.

were caught in Station 9 on the southern shore near the mouth of Tampa Bay (Table 1, Figure 1). Red drum were most frequently collected on fishing trips to Stations 1 and 3 (77.3% and 65.8%, respectively; Table 1). In all stations, red drum were more often caught with circle hooks than with J–hooks (Ratio of circle hook to J–hook caught fish ranged from 2.3 – 14.0; Table 1), and the size distribution of red drum captured with J–hooks differed significantly than those caught with circle hooks (K_{sa} = 2.37, p < 0.0001), although mean lengths differed minimally (346.4 and 328.5 mm SL, respectively). Most red drum were captured with natural baits (n = 1,359), primarily live shrimp (n = 800), followed by dead natural bait (n = 360) and live fish (n = 196). The remaining 46 red drum were captured with artificial lures.

Red drum collected in monthly hook–and–line sampling ranged from 135 to 680 mm SL (Figure 2). Catch was dominated by sublegal red drum (< 379 mm SL, n = 994, 70.8%) but also included legal slot–size red drum (379–570 mm SL, n = 377, 26.8%) and a few individuals larger than the legal slot–size (> 570 mm SL, n = 34, 2.4%; Figure 2). Sizes of red drum varied significantly among fishing stations (X^2 = 709.97, p < 0.0001;

TABLE 2. Summary of water quality (mean, standard error (se), range) and dominant habitat characteristics in hook-and-line sampling stations in Tampa Bay, FL (April 2005 – December 2007). Dominant habitat types listed are those that had the highest proportion of sites (in parentheses) in each station in which fishing was targeted towards that habitat type. SAV—submerged aquatic vegetation.

Station	Temperature (°C) Mean ± se	Range	Dissolved oxygen (mg/l) Mean ± se	Range	Salinity Mean ± se	Range	Dominant habitat types (Proportion of sites) Shore type	Bottom Vegetation	Substrate
1	25.4 ± 0.3	14.0 - 34.4	7.5 ± 0.2	0.6 - 14.5	27.2 ± 0.2	13.0 - 33.6	Overhanging (0.54)	SAV (0.62)	Sand (0.69)
2	25.4 ± 0.3	16.1 - 34.2	6.7 ± 0.1	1.8 - 11.8	26.0 ± 0.3	8.6 - 31.6	Overhanging (0.64)	None (0.68)	Sand (0.53)
3	24.6 ± 0.3	14.9 - 32.7	6.5 ± 0.1	0.4 - 13.0	21.6 ± 0.5	0.1 - 34.0	Overhanging (0.83)	None (0.69)	Mud (0.53)
4	25.1 ± 0.3	16.5 - 40.0	6.4 ± 0.1	1.7 - 13.0	24.1 ± 0.4	6.1 - 32.5	Overhanging (0.74)	None (0.65)	Mud (0.54)
5	25.2 ± 0.3	13.8 - 33.3	6.0 ± 0.2	0.6 - 10.6	26.4 ± 0.2	13.7 - 35.0	Overhanging (0.76)	None (0.53)	Mud (0.60)
6	25.0 ± 0.3	15.0 - 31.6	6.2 ± 0.1	2.4 - 11.6	16.7 ± 0.5	0.1 - 30.4	Overhanging (0.80)	None (0.62)	Sand (0.45)
7	24.6 ± 0.3	16.0 - 33.0	5.9 ± 0.2	0.6 - 13.9	28.8 ± 0.2	21.7 - 37.8	Overhanging (0.91)	SAV (0.57)	Sand (0.74)
8	25.1 ± 0.3	13.1 - 32.6	6.7 ± 0.2	1.6 - 13.4	31.2 ± 0.2	23.3 - 36.7	Overhanging (0.75)	SAV (0.70)	Sand (0.68)
9	25.3 ± 0.3	15.8 - 32.1	6.0 ± 0.1	1.0 - 11.4	30.8 ± 0.3	10.0 - 36.0	Overhanging (0.81)	SAV (0.75)	Sand (0.81)
10	25.1 ± 0.4	12.0 - 32.3	6.4 ± 0.1	1.6 - 12.4	33.3 ± 0.1	20.6 - 36.5	Overhanging (0.71)	SAV (0.71)	Sand (0.69)

TABLE 3. Hook positions for red drum captured in Tampa Bay, FL during monthly hook-and-line sampling (April 2005–December 2007) and mortality experiments (November 2005–March 2008). The percentage of the total catch by hook position is given in parentheses for each sampling type.

Hook position	Number caught (%)	
	Monthly hook-and-line sampling	Mortality experiments
Lip	1,199 (85.3%)	200 (79.7%)
Inside mouth	45 (3.2%)	19 (7.5%)
Throat	60 (4.3%)	16 (6.4%)
Gut	93 (6.6%)	16 (6.4%)
Other	8 (0.6%)	0
Total	**1,405**	**251**

Figure 2). Sublegal red drum were collected in every station, but were more commonly caught in areas of the bay near tidal rivers and creeks. These backwater habitats consisted principally of mangroves (overhanging vegetation), soft substrates, and oyster bars (Stations 2 – 6, Figures 1 and 2, Table 2). On average, the smallest red drum were caught in Stations 4 and 5 (mean SL = 253 and 254 mm, respectively; Figure 2). Fewer slot–size fish (379–570 mm SL) were caught in less saline areas of the bay (Stations 3-6, Table 2). On average, larger red drum were collected predominantly from seagrass flats and sandy substrates near either the interbay peninsula (Station 1) or the mouth of the bay (Stations 9 and 10, Figure 2, Table 2).

Most red drum (n = 1,244) were hooked in shallow anatomical locations such as the lip or inside the mouth (Table 3). Only 153 fish were recorded as either gut–hooked or throat–hooked. Ninety percent of all red drum caught during monthly sampling were released in good condition. Of the 153 fish hooked in deep anatomical locations, 22.9% (n = 35) were released in fair, poor, or dead condition, whereas 0.8% (n = 11) of the 1,244 fish hooked in shallow anatomical locations were released in fair or poor condition, and none were dead upon release.

Catch–and–release mortality experiments

A total of 251 red drum (range: 203–618 mm SL; Table 4, Figure 3) were caught during 9 catch–and–release mortality experiments; 14 of these fish died during the 48–hour holding period. The overall mortality rate for all experiments combined was 5.6% (Table 4). Water temperature and hook position were correlated with the probability of mortality (Table 5), and the logistic model exhibited acceptable goodness–of–fit (Hosmer–Lemeshow test, X^2 = 2.46, p = 0.87). Of environmental variables,

only water temperature was significantly correlated with red drum mortality (Table 5). Eight of the 14 mortalities (57%) occurred in water temperatures > 26°C (Figure 4). Hook position was also significantly associated with red drum mortality (Table 5). Lip–hooked fish had the lowest short–term mortality rate (3.5%), whereas fish hooked in the throat had the highest rate (18.8%; Figure 5). Hook type was not directly associated with mortality; however, significantly more red drum were deep–hooked (in the throat or gut) when J–hooks were used instead of circle hooks (Table 5, Figure 6).

A variety of parameters that we expected to influence mortality were not significant in the logistic regression model. Release condition was not significantly associated with short–term mortality probably due to the fact that 97% of the red drum caught during mortality experiments were released in good condition. Similar to what was found in the monthly fishing experiments, 18.8% (n = 6) of the 32 deep–hooked fish were released in fair or poor condition, whereas only 0.9% (n = 2) of the 219 shallow–hooked fish were released in fair condition. The remainder of the fish were released in good condition. Whether the hook was removed also did not contribute significantly to mortality; however, in our study, hooks were only left in gut–hooked or throat–hooked fish (75% and
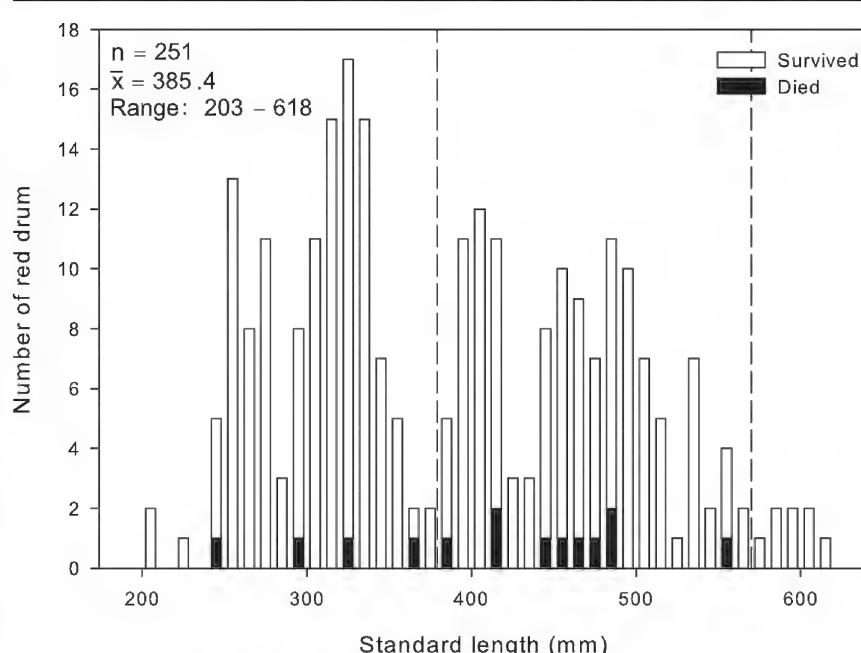

Figure 3. Length frequency of red drum (by 10 mm size bins) collected during catch-and-release mortality experiments in Tampa Bay, November 2005–March 2008. White bars represent fish that survived to be released after the 48 h holding period; black bars represent individuals that died during the experiment. The Florida legal slot size limit, estimated from SL/TL length regressions, is indicated by vertical dashed lines on each plot.

TABLE 4. *Summary of the number and sizes of red drum caught during each catch-and-release mortality experiment conducted in Tampa Bay, FL from 2005–2008, and those that were eventually recaptured from 2005–2012. Percentages were not calculated for experiments in which < 10 red drum were collected.*

Month	Station(s)	SL (mm, mean ± se)	# caught	# died	% died	# recaptured	% recaptured
Nov. 2005	3, 4	275.8 ± 5.4	52	3	5.8	5	10.2
Dec. 2005	3, 4	306.1 ± 5.7	17	0	0.0	2	11.8
Mar. 2006	1	379.0 ± 9.2	68	2	2.9	5	7.6
July 2006	1	476.4 ± 16.2	19	2	10.5	4	23.5
Nov. 2006	3, 4	355.2 ± 26.0	12	0	0.0	1	8.3
May 2007	1	495.7 ± 11.4	26	0	0.0	0	0.0
Aug. 2007	1	515.5 ± 19.7	6	0	–	0	–
Oct. 2007	10	435.5 ± 5.2	34	6	17.6	4	14.3
Mar. 2008[a]	1	431.2 ± 19.4	17	1	5.9	1	6.3
Total		**385.4 ± 6.1**	**251**	**14**	**5.6**	**22**	**9.3**

[a]Additional experiment conducted after monthly hook-and-line sampling ended.

81% of fish captured with each hook position, respectively) including 3 of the fish that died (1 gut–hooked, 2 throat–hooked). Hooks were over 3 times more likely to be removed from fish caught with circle hooks than those caught with J–hooks (5.4% and 18.6% of fish did not have hooks removed, respectively). Tagging did not significantly contribute to red drum mortality. One of the 39 untagged control fish died within the 48 h holding period, resulting in a mortality rate of 2.6%. Other factors such as handling time, fight time, dissolved oxygen, and fish length were not significantly correlated with mortality or significantly different between fish that died and survived (all ANOVAS p > 0.05). Although fish length was not significantly correlated with the probability of mortality, most red drum that died were of legal size (n = 10, Figure 3), whereas only 4 of those smaller than the slot size and none that were larger than the slot size died.

Necropsies of red drum that died during mortality experiments revealed 3 internal conditions: no noticeable injury, a torn esophagus, or an injury to the heart (Table 6). Red drum caught on circle hooks more frequently had no noticeable injury. Of the 9 individuals with no noticeable injury, 6 had been shallow–hooked and caught in experiments with warm water temperatures (July 2006 and October 2007; Figure 4). Red drum mortalities with injuries to the esophagus or heart had mostly been caught with J–hooks (n = 4; Table 6); only one had been caught with a circle hook. Based on tag recaptures through October 2012, relative long–term survival of red drum released alive (n = 237) was extremely good after accounting for short–term mortality through mortal-ity experiments (Table 7). An overall recapture rate of 9% indicated high survival of fish that did not suffer short–term mortality, and those fish hooked in deeper anatomical locations had a lower relative survival than those hooked in shallower locations. Relative survival was not markedly lower for red drum captured with J–hooks versus circle hooks and reinforces the short–term catch–and release results that indicated that hook type was not a significant factor in mortality. The confidence intervals were extremely broad for relative survival rates by hook position, reflecting the low sample size and rarity of instances in which fish were not lip–hooked. However, for hook position and type, the results suggest that
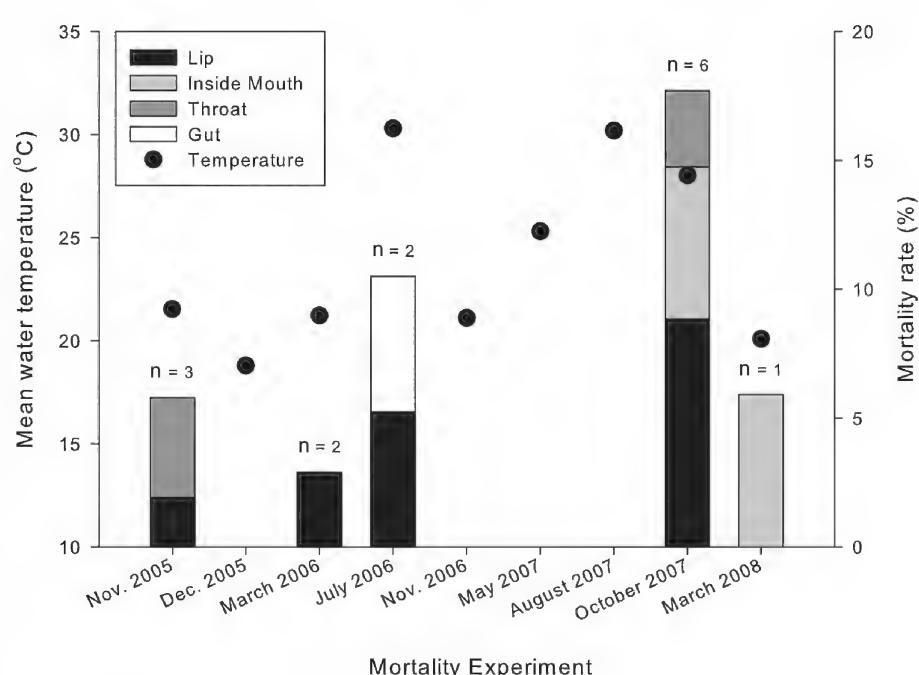

Figure 4. *Mean water temperature recorded at the main holding pen and percent mortality of red drum during catch-and-release mortality experiments, 2005–2008. Black dots represent the mean temperature. Stacked bars represent the percentage of red drum that died during each experiment and the hook position of each fish. n = number of red drum mortalities for a given experiment.*

TABLE 5. *Significant Wald Chi-square statistics (X^2) from 2 different logistic regression models (forward selection at p < 0.05) describing the factors associated with the probability of red drum mortality and deep-hooking, respectively.*

Response	Factor	df	X^2	p value
Mortality	Temperature (°C)	1	4.9786	0.0257
	Hook position	3	8.6198	0.0348
Deep-hooking	Hook type	1	11.7849	0.0006

the risk ratios are greater than 1, indicating that there is a greater chance of recapture for fish that were lip—hooked or caught by circle hooks. Recapture rates varied over the different mortality experiments and did not indicate a seasonal (or water temperature related) trend (Table 4). Interestingly, the highest long—term recapture rates were associated with the experiments that had the highest mortality rates (July 2006 and October 2007).

Discussion

Spatial differences were evident with respect to the size structure of red drum collected during this study; these differences generally reflected ontogenetic patterns of habitat use by various life—history stages of red drum. Small (sub—legal) red drum were collected primarily from areas near tidal rivers and small tidal creeks. These backwater, less saline habitats contained soft substrates and oyster bars adjacent to mangrove shorelines that represent the preferred habitats of young—of—year and small sub—legal red drum (Peters and McMichael 1987, Stunz et al. 2002a). In contrast, legal—sized red drum were most frequently collected near the interbay peninsula (Station 1; Figures 1 and 2) and to a lesser extent near the mouth of the Tampa Bay estuary (Stations 9 and 10) which are characterized by large expanses of seagrass flats with sandy substrates. The interbay peninsula contains a functional large marine protected area because boaters are not allowed in the security zone surrounding MacDill Air Force Base, which encompasses the southern tip of the interbay peninsula. These restrictions on boaters may offer some protection for legal—sized red drum in this area of the bay. The large red drum caught in the lower portion of Tampa Bay may be individuals that were staging prior to emigration into nearshore GOM waters (Switzer et al. 2009). With the exception of the larger red drum caught around the interbay peninsula, our length—frequency and ancillary tag—recapture data (Switzer et al. 2009) indicate that red drum move from the primary nursery areas to the mouth of the bay as they grow.

The survival of red drum released after being caught by an angler has significant bearing on the inshore population and ultimately influences the numbers of red drum that emigrate into nearshore GOM waters. Due to recent interest in reopening Federal waters of the GOM for a limited harvest of large adult red drum, the catch—and—release survival of red drum within the estuary and their rate of escapement could be relevant to this discussion. The short—term catch—and—release mortality rate calculated for red drum in this study is similar to that seen in other studies, which reported mortality rates < 10% throughout waters adjacent to the southeastern United States (Matlock et al. 1993, Muoneke and Childress 1994, Aguilar et al. 2002, Vecchio and Wenner 2007). Overall, the catch—and—release mortality rate is low for red drum; however, we have shown that this rate can be significantly elevated with deep—hooking and with higher water temperature. Long—term survival rates estimated from tag return data were also quite high, and although these rates were extremely variable, the results correspond to conclusions regarding differential survival among hook positions.

Differences in catch—and—release mortality rates related to the anatomical location of the hook have been reported in several studies (Muoneke and Childress 1994, Aguilar et al. 2002, Aalbers et al. 2004, Bartholomew and Bohnsack 2005, Cooke et al. 2005, Vecchio and Wenner 2007). The major-
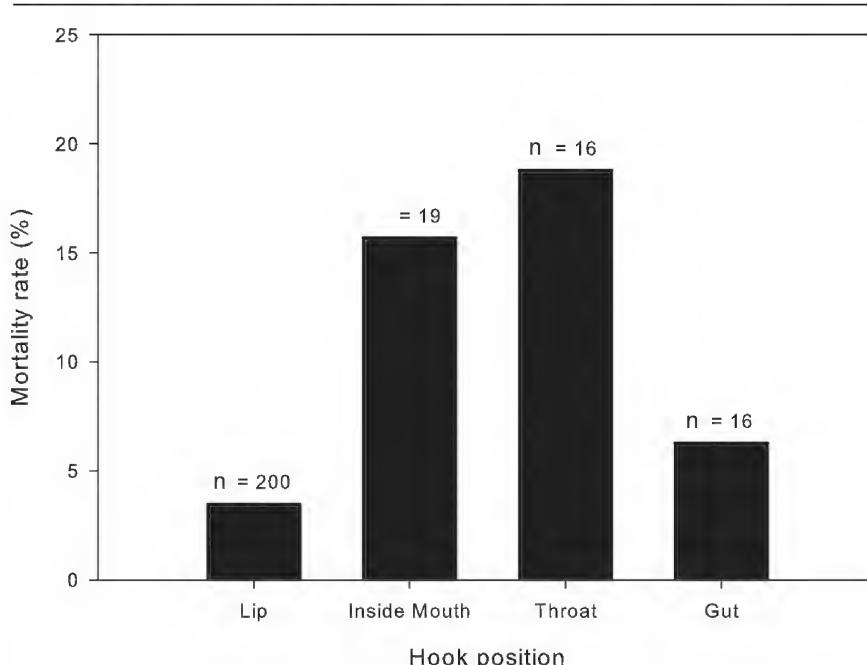

Figure 5. *Percentage of red drum that died during catch-and-release mortality experiments by hook position (lip = corner of mouth; inside mouth = buccal cavity; throat = anterior of pharyngeal teeth; gut = posterior of pharyngeal teeth). n = number of red drum hooked in a given anatomical location.*
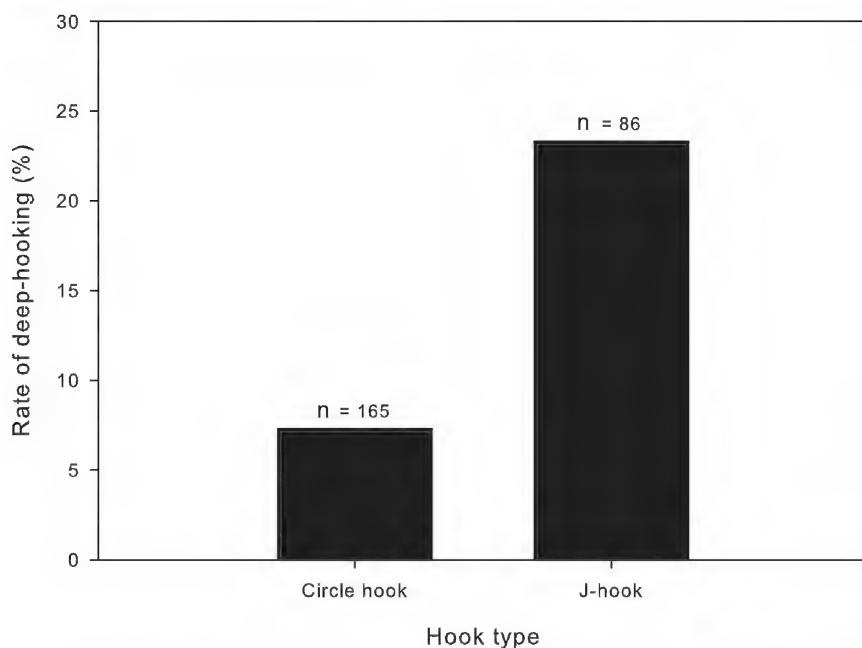
Figure 6. Percentage of red drum deep-hooked (throat- or gut-hooked), by hook type, during catch-and-release mortality experiments. n = number of red drum captured using each hook type.

ity of red drum caught in our study were hooked in shallow anatomical locations such as the lip or inside the mouth; hook wounds in these areas usually result in fewer injuries to vital organs, resulting in a lower rate of short—term mortality for the total population (Aalbers et al. 2004, Cooke and Suski 2004, Vecchio and Wenner 2007). Conversely, hook wounds in deep anatomical locations such as the gills, esophagus, or gut have been found to damage vital organs and contribute to higher rates of mortality (Muoneke and Childress 1994). During monthly hook—and—line sampling and mortality experiments, the percentage of deep—hooked red drum released in fair or poor condition was greater than that of shallow—hooked fish. In addition, 3 of the 4 deep—hooked fish that died during mortality experiments had noticeable injuries either to the esophagus or to the heart. Of all deep—hooked fish in our study, fish that were hooked in the throat had the highest mortality rate, and gut—hooked fish had the lowest mortality rate. This lower mortality rate for gut—hooked fish may be a result of the small sample size of these fish in our study (n = 16); however, the percentage of gut—hooked fish (6.4%) is consistent with that seen in our monthly hook—and—line sampling (6.6%), and the sample size is equivalent to that of throat—hooked fish. Fish that are hooked in the throat or gill region often sustain immediate trauma and bleeding, which is reflected in the higher mortality rates observed. In contrast, adverse effects due to gut—hooking may not be apparent in the short—term, especially if the hook is left inside the fish, which is often the case in order to avoid additional trauma. The removal of deeply

ingested hooks can increase handling stress, cause significant trauma, and contribute to short—term mortality (Muoneke and Childress 1994); however, hooks left embedded in the peritoneal cavity or gut may cause damage that can contribute to long—term (> 48 h) post—release mortality (Lawson and Sampson 1996, Aalbers et al. 2004, Vecchio 2006). Of the fish that survive, hooks may dissolve within the fish, be extruded, or show evidence of tissue growth around the wounds (Muoneke and Childress 1994). Although recapture rates of red drum based on long—term tag return data were high and variable for all hook positions, these rates were lowest for gut—hooked fish suggesting long—term survival may be affected.

Release condition was not associated with mortality in this study; however, in other species, the amount of bleeding (Aalbers et al. 2004, Fabrizio et al. 2008, Grixti et al. 2008) and general release condition (Burns et al. 2008; Sumpton et al. 2008) has been associated with mortality or used for tag—recapture mortality estimates, respectively. Similar to our study, Aguilar et al. (2002) observed very few red drum that exhibited external bleeding. The vast majority of the red drum caught during monthly hook—and—line sampling and mortality experiments were released in good condition (90% and 97%, respectively), which indicates that this may not be a good indicator of short—term mortality. Several caveats must be presented with the results of net pen studies of fish survival. Confinement studies generally preclude large scale ecosystem interactions, like predation, and do not assess behavioral movements of fish after the catch—and—release event (Donaldson et al. 2008). Also, confinement in pens may possibly increase stress and mortality. Although a fish may be released in good condition, this assessment is subjective and may not reflect the physiological stress of being captured.

Contrary to other published work (Bartholomew and

TABLE 6. Number of red drum mortalities by type of injury, hook position, and hook type as determined by necropsy. Three main observations were recorded: no visible injury; injury to the esophagus; and injury to the heart. Hook type with which the individuals were captured is represented by C (circle hook) or J (J-hook).

	No injury		Esophagus		Heart		
Hook position	**C**	**J**	**C**	**J**	**C**	**J**	**Total**
Lip	5	2	–	–	–	–	7
Inside mouth	–	1	–	1	–	1	3
Throat	1	–	–	2	–	–	3
Gut	–	–	–	–	1	–	1
Totals	**6**	**3**	**—**	**3**	**1**	**1**	**14**

TABLE 7. *Long-term relative survival rates of red drum released alive after mortality experiments based on tag recaptures from April 2005 – October 2012 by hook position and hook type. Relative survival (S) and 95% confidence interval (CI) calculations for other categories were based on the assumption that fish that were lip-hooked or captured by circle hook had a survival rate of 1.*

Variable	Number of red drum Released	Number of red drum Recaptured	Recapture rate (%)	S	CI
Hook position					
Lip	193	17	8.81	1.00	–
Inside mouth	16	3	18.75	2.13	0.70–6.50
Throat	13	1	7.69	0.87	0.13–6.06
Gut	15	1	6.67	0.76	0.11–5.30
Total	**237**	**22**	**9.28**	**–**	**–**
Hook type					
Circle	158	15	9.49	1.00	–
J	79	7	8.86	0.93	0.40–2.20
Total	**237**	**22**	**9.28**	**–**	**–**

Bohnsack 2005, Beckwith and Rand 2005, Jones 2005, Vecchio and Wenner 2007), hook type was not directly associated with mortality in this study. Similar to other studies, however, a higher proportion of red drum were captured and deep–hooking was less frequent with circle hooks as opposed to J–hooks (McEachron et al. 1985, Vecchio and Wenner 2007). Deep–hooking is more common with J–hooks (Bartholomew and Bohnsack 2005, Beckwith and Rand 2005, Jones 2005, Vecchio and Wenner 2007) because their design allows them to catch on tissue in the gut or throat, whereas the shape of a circle hook makes it more likely to hook the lip after bypassing the esophageal tissues without penetrating them. Hook trauma was noticeable in several mortalities involving J–hooks; 4 of the 5 fish that died and had injuries to the esophagus or heart were caught on J–hooks. Because of the tendency of J–hooks to become deeply embedded in tissue, it is possible that some red drum may have broken free with embedded J–hooks before being landed, therefore decreasing observed catch and making their survival unknown (Vecchio and Wenner 2007). In addition to hook type, hook size has been associated with mortality rates (Muoneke and Childress 1994) and is related to minimum capture size in some species (Otway and Craig 1993, Cooke et al. 2005). A study of red drum, however, showed no relationship between hook size and size of fish caught (Aguilar et al. 2002). Because we used small J–hooks and circle hooks of the same gauge (1/0) in this study, future research could focus on the differences related to hook size in deep–hooking, mortality, or catch rates of different sized fish.

Although water temperature was significantly correlated with mortality in this study, in other studies in GOM estuaries this association was variable. A study of spotted seatrout, *Cynoscion nebulosus,* in Florida detected no significant correlation between catch–and–release mortality and water temperature (Murphy et al. 1995), but the reverse was true for spotted seatrout in Texas (James et al. 2007). The catch–and–release mortality of common snook, *Centropomus undecimalis,* was not significantly related to temperature (Taylor et al. 2001), which may be explained by the ability of this tropical species to tolerate and spawn in temperatures warmer than 30°C. Studies of red drum and other species in various locations have shown positive correlations between water temperature and mortality (Muoneke and Childress 1994, Schisler and Bergersen 1996, Latour et al. 2001, Bartholomew and Bohnsack 2005). Latour et al. (2001) observed a relatively high mortality rate (19.1%) for large red drum (>550 mm SL) in water temperatures above 25°C, while Aguilar et al. (2002) found no correlation of red drum mortality with temperature. In our experiments, several red drum without noticeable injury died in water warmer than 26°C, suggesting that thermal stress may be a contributing factor to mortality. Catch–and–release mortality rates calculated in this study represent the most southerly such estimates for red drum and represent a broader range of water temperatures than found in most other studies (Matlock et al. 1993, Muoneke and Childress 1994). Red drum have been observed in waters at temperatures from 2 to 33°C (Mercer 1984), but movement from warm, shallow waters to cooler, deeper waters is commonly observed. The inability of red drum to escape into waters deeper (and cooler) than the holding pens may have added to thermal stress and contributed to mortality rates during experiments in warmer waters. The upper limits of thermal tolerance for juvenile red drum (71–155 mm SL) from hatcheries in Texas and South Carolina ranged from 28.8 to 35.7°C, depending on acclimation temperature (Procarione and King 1993). In our study, the greatest mortality rate was observed during an experiment in October 2007, when mean water temperature was 28.0°C, although a large proportion of these fish survived long–term as suggested by the high recapture rate. Contrary to expectations, all 6 fish caught during an experiment in August 2007, when the mean water temperature was 30.2°C, survived. These fish may have experienced less stress, since only 6 fish were in the holding pen and all had been shallow–hooked with circle hooks. None of these fish were recaptured, but this is not surprising considering the small sample size. Although the level of DO was not correlated with mortality in our study, low DO levels in high–temperature waters may contribute to mortality because of increased respiratory demands. Injuries sustained during catch–and–release fishing may also be more prone to infection in warmer waters (Muoneke 1992).

Stress associated with increasing fight and handling times did not affect the mortality of red drum in this study. Re-

38

sults from prior studies examining these angling–related factors have been equivocal. For example, a rainbow trout, *Oncorhynchus mykiss*, study showed an increase in probability of mortality with increased fight time and handling time out of the water (Schisler and Bergersen 1996). However, a study on chinook salmon, *Oncorhynchus tshawytscha*, did not have higher mortality rates associated with increased stress (Wertheimer et al. 1989). Responsible fighting and handling techniques were practiced during our study, including fishing actively, limiting play in the line while fighting the fish, supporting the fish with both hands, processing the fish as quickly as possible, and leaving the hook in deeply hooked fish (Bartholomew and Bohnsack 2005). On average, our fight times (< 1 minute) and handling times (< 3 minutes) were in a conservative range and probably did not cause critical stress levels that would be expected to increase mortality. Thus, our catch–and–release mortality estimates are conservative with regard to these factors.

Previous studies have found no relationship between fish size and catch–and–release mortality (Muoneke and Childress 1994, Aguilar et al. 2002, Stunz and McKee 2006). Although fish length was not a significant factor in this study, a slightly larger percentage of legal–sized red drum died than sublegal or supralegal red drum. This potential difference in mortality rates of legal–sized red drum should be investigated further and may have implications for management. The size range of fish captured during our catch–and–release mortality experiments was similar to that observed in our monthly hook–and–line trips, which represented a relatively broad size range of red drum available to the recreational fishery throughout Tampa Bay. Because we caught relatively few supralegal fish, however, future research efforts should focus on either large subadult or young adult red drum, which often school in the lower estuary in the fall. Schools of these sizes of red drum are common in shallow estuarine waters during the warmest months, when DO levels are lowest. These trophy–size red drum are heavily targeted by fishing guides and recreational anglers and may be subject to longer fight and handling times because of their size. This heavy fishing pressure was potentially reflected by the high mortality coupled with a high recapture rate of fish from the October 2007 experiment conducted in this area. In addition, all but one of our mortality experiments were conducted in the upper bay and may not be representative of the population in the lower bay. Although in our study no fish larger than legal size died within the 48 h holding period, these factors could very well contribute to some short–term catch–and–release mortality.

The results of this study can be used in stock assessments and outreach programs aimed at recreational anglers. In addition to calculating catch–and–release mortality estimates for the southerly Tampa Bay estuary, this study incorporates data collected from year–round hook–and–line sampling for red drum to estimate sizes available to the recreational fishery. The most recent Florida Fish and Wildlife Conservation Commission (FWC) red drum stock assessment (Murphy and Munyandorero 2008) assumed 5% post–release mortality, which is closely in line with our estimate. Depending on water temperature and hook position, however, the probability of mortality may be substantially higher. The FWC stock assessment also concluded that a 30% escapement goal was barely being met as of 2007 (Murphy and Munyandorero 2008). If the number of anglers continues to increase, as it has for the past several decades, this goal will no longer be met under current regulations coupled with the 5% post–release mortality assumption (Murphy and Munyandorero 2008). One option would be to support outreach efforts that educate anglers on the advantages of using circle hooks. Our study found that anglers using circle hooks deep–hooked fish about one–third as often as those using J–hooks, which corresponded to a lower incidence of hook trauma. Catch–and–release fishing is an effective management tool for reducing take in the red drum fishery and should be encouraged as a management strategy; however, the practice can contribute to cryptic mortality, especially with heavy fishing pressures, and these impacts should not be overlooked when evaluating the overall health of fish populations.

Literature Cited

Aalbers, S.A., G.M. Stutzer, and M.A.Drawbridge. 2004. The effects of catch–and–release angling on the growth and survival of juvenile white seabass captured on offset circle and J–type hooks. North American Journal of Fisheries Management 24:793-800.

Aguilar, R., P.S. Rand, and G.H. Beckwith. 2002. Quantifying the catch–and–release mortality rate of adult red drum in the Neuse River Estuary. 01–FEG–07. Final Report. North Carolina Fisheries Resource Grant Program, Raleigh, NC, USA, 44 p.

Bacheler, N.M., L.M. Paramore, J.A. Buckel, and F.S. Scharf. 2008. Recruitment of juvenile red drum in North Carolina: spatiotemporal patterns of year–class strenght and validation of a seine survey. North American Journal of Fisheries Management 28:1086-1098.

Baltz, D M., J W. Fleeger, C.F. Rakocinski, and J.N. McCall. 1998. Food, density, and microhabitat: factors affecting growth and recruitment potential of juvenile saltmarsh fishes. Environmental Biology of Fishes 53:89-103.

Bartholomew, A. and J.A. Bohnsack. 2005. A review of catch–and–release angling mortality with implications for no–take reserves. Reviews in Fish Biology and Fisheries 15:129-154.

Beckwith, G.H. and P.S. Rand. 2005. Large circle hooks and short leaders with fixed weights reduce incidence of deep hooking in angled red drum. Fisheries Research 71:115-120.

Bugley, K. and G. Shepherd. 1991. Effect of catch–and–release angling on the survival of black sea bass. North American Journal of Fisheries Management 11:468-471.

Burns, K.M., N.J. Brown–Peterson, R.M. Overstreet, J. Gannon, P. Simmons, J. Sprinkle, and C. Weaver. 2008. Evaluation of the efficacy of the current minimum size regulations for selected reef fish based on release mortality and fish physiology. Technical Report No. 1176. Mote Marine Laboratory, Sarasota, FL, USA, 75 p.

Cooke, S.J., B.L. Barthel, C.D. Suski, M.J. Siepker, and D.P. Philipp. 2005. Influence of circle hook size on hooking efficiency, injury, and size selectivity of bluegill with comments on circle hook conservation benefits in recreational fisheries. North American Journal of Fisheries Management 25:211-219.

Cooke, S.J. and C.D. Suski. 2004. Are circle hooks an effective tool for conserving marine and freshwater recreational catch–and–release fisheries? Aquatic Conservation: Marine and Freshwater Ecosystems 14:299-326.

Daniel, L.B. 1988. Aspects of the biology of juvenile red drum, *Sciaenops ocellatus*, and spotted seatrout, *Cynoscion nebulosus* (Pisces: Sciaenidae), in South Carolina. MS thesis, College of Charleston, Charleston, SC, USA, 116 p.

Donaldson, M.R., R. Arlinghaus, K.D. Hanson, and S.J. Cooke. 2008. Enhancing catch–and–release science with biotelemetry. Fish and Fisheries 9:79-105.

Fabrizio, M.C., F.S. Scharf., G.R.Shepherd, and J.E. Rosendale. 2008. Factors affecting catch and release mortality of bluefish. North American Journal of Fisheries Management 25:533–536.

Grixti, D., S.D. Conron, A. Morison, and P.L. Jones. 2008. Estimating post–release survival and the influential factors for recreationally caught Black bream (*Acanthopagrus butcheri*) in the Glenelg River, south eastern Australia. Fisheries Research 92:303–313.

Hueter, R.E., C.A. Manire, J.P. Tyminski, J.M. Hoenig, and D.A. Hepworth. 2006. Assessing mortality of released or discarded fish using a logistic model of relative survival derived from tagging data. Transactions of the American Fisheries Society 135: 500–508.

James, J.T., G.W. Stunz, D.A. McKee, and R.R. Vega. 2007. Catch–and–release mortality of spotted seatrout in Texas: effects of tournaments, seasonality, and anatomical hooking location. North American Journal of Fisheries Management 27:900-907.

Jones, T.S. 2005. The influence of circle hooks on the capture efficiency and injury rate of walleyes. North American Journal of Fisheries Management 25:725-731.

Latour, R.J., K.H. Pollock, C.A. Wenner, and J.M. Hoenig. 2001. Estimates of fishing and natural mortality for subadult red drum in South Carolina waters. North American Journal of Fisheries Management 21:733-744.

Lawson, P.W. and D.B. Sampson. 1996. Gear–related mortality in selective fisheries for ocean salmon. North American Journal of Fisheries Management 16:512-520.

Matlock, G.C., L.W. McEachron, J.A. Dailey, P.A. Unger, and P. Chai. 1993. Short–term hooking mortalities of red drums and spotted seatrout caught on single barb and treble hooks. North American Journal of Fisheries Management 13:186-189.

McEachron, L.W., A.W. Green, G.C. Matlock, and G.E Saul. 1985. A comparison of trotline catches on two hook types in the Laguna Madre. Management Data Series No. 86. Texas Parks and Wildlife Department, Austin, TX, USA, 44 p.

McMichael, Jr., R.H. 2010. Fisheries—Independent Monitoring Program 2009 Annual Data Summary Report. In—house Report, IHR—2010—001, Fish and Wildlife Research Institute. Florida Fish and Wildlife Conservation Commission, St. Petersburg, FL, USA, 340 p.

Mercer, L.P. 1984. A biological and fisheries profile of red drum, *Sciaenops ocellatus*. Special Science Report 41. Community Development, Division of Marine Fisheries, North Carolina Department of Natural Resources, Raleigh, NC, USA, 89 p.

Muoneke, M.I. 1992. Seasonal hooking mortality of bluegills caught on natural bait. North American Journal of Fisheries Management 12:645–649.

Muoneke, M.I. and W.M. Childress. 1994. Hooking mortality: a review for recreational fisheries. Reviews in Fisheries Science 2:123–156.

Murphy, M.D. and R.E. Crabtree. 2001. Change in the age structure of nearshore adult red drum off west—central Florida related to recruitment and fishing mortality. North American Journal of Fisheries Management 21:671–678.

Murphy, M.D. and J. Munyandorero. 2008. An assessment of the status of red drum in Florida waters through 2007. IHR 2008—008. Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute, St. Petersburg, FL, USA, 106 p.

Murphy, M.D. and R.G. Taylor. 1990. Reproduction, growth, and mortality of red drum *Sciaenops ocellatus* in Florida waters. Fishery Bulletin 88:531–542.

Murphy, M.D., R.F. Heagey, V.H. Neugenbauer, M.D. Gordon, and J.L. Hintz. 1995. Mortality of spotted seatrout released from gill net or hook—and—line gear in Florida. North American Journal of Fisheries Management 15:748–753.

National Oceanographic and Atmospheric Administration (NOAA). 2008. Recreational fisheries statistics queries. Available: www.st.nmfs.noaa.gov/st1/recreational/queries/index.html. (viewed 1/28/2013)

Otway, N.M. and J.R. Craig. 1993. Effects of hook size on catches of undersized snapper *Pagrus auratus*. Marine Ecology Progress Series 93:9–15.

Peters, K.M. and R.H. McMichael, Jr. 1987. Early life history of the red drum, *Sciaenops ocellatus* (Pisces: Sciaenidae), in Tampa Bay, Florida. Estuaries 10:92–107.

Procarione, L.S. and T.L. King. 1993. Upper and lower temperature tolerance limits for juvenile red drums from Texas and South Carolina. Journal of Aquatic Animal Health 5:208–212.

SAS Institute Inc. 2006. Base SAS® 9.1.3 procedures guide, 2nd edition, Volumes 1-4. SAS Institute Inc., Cary, NC, USA, 1,906 p.

Schisler, G.J. and E.P. Bergersen. 1996. Postrelease hooking mortality of rainbow trout caught on scented artificial baits. North American Journal of Fisheries Management 16:570–578.

Sokal, R.R. and F. J. Rohlf. 1995. Biometry, 3rd ed. W.H. Freeman and Company, New York, NY, USA, 887 p.

Stunz, G.W., T.J. Minello, and P.S. Levin. 2002a. A comparison of early juvenile red drum densities among various habitat types in Galveston Bay, Texas. Estuaries 25:76–85.

Stunz, G.W., T.J. Minello, and P.S. Levin. 2002b. Growth of newly settled red drum *Sciaenops ocellatus* in different estuarine habitat types. Marine Ecology Progress Series 238:227–236.

Stunz, G.W. and D.A. McKee. 2006. Catch—and—release mortality of spotted seatrout in Texas. North American Journal of Fisheries Management 26:843–848.

Sumpton, W., D. Mayer, I. Brown, B. Sawynok, M. McLennan, A. Butcher, and J. Kirkwood. 2008. Investigation of movement and factors influencing post—release survival of line—caught coral reef fish using recreational tag—recapture data. Fisheries Research 92:189—195.

Switzer, T.S., M.D. Murphy, B.L. Winner, K.E. Flaherty, and J.L. Vecchio. 2009. Size and age structure and catch—and—release mortality estimates of sub—adult and adult red drum (*Sciaenops ocellatus*) in the Tampa Bay estuary and nearshore Gulf of Mexico waters. NOAA/NMFS MARFIN Final Report (Grant # NA05NMF4331078). St. Petersburg, FL, USA, 71 p.

Taylor, R.G., J.A. Whittington, and D.E. Haymans. 2001. Catch—and—release mortality rates of common snook in Florida. North American Journal of Fisheries Management 21:70–75.

United States Geological Survey. 2008. Tampa Bay study. Available: gulfsci.usgs.gov/tampabay/index.html. (viewed 1/28/2013).

Vecchio, J.L. 2006. Catch and release in South Carolina's red drum recreational fishery. MS thesis. College of Charleston, Charleston, SC, USA, 242 p.

Vecchio, J.L. and C.A. Wenner. 2007. Catch—and—release mortality in subadult and adult red drum captured with popular fishing hook types. North American Journal of Fisheries Management 27:891–899.

Wenner, C.A. 1992. Red drum: natural history and fishing techniques in South Carolina. Report 17. South Carolina Department of Natural Resources, Marine Resources Division, Charleston, SC, USA, 45 p.

Wertheimer, A., A. Celewycz., H. Jaenicke., D. Mortensen, and J. Orsi. 1989. Size—related hooking mortality of incidentally caught chinook salmon, *Oncorhynchus tshawytscha*. Marine Fisheries Review 51:28—35

Yokel, B. J. 1966. A contribution to the biology and distribution of red drum, *Sciaenops ocellata*. MS thesis, University of Miami, Miami, FL, USA, 166 p.

2013

Characterization of Ichthyoplankton in the Northeastern Gulf of Mexico from Seamap Plankton Surveys, 1982-1999

Joanne Lyczkowski-Shultz
National Oceanic and Atmospheric Administration

David S. Hanisko
National Oceanic and Atmospheric Administration

Kenneth J. Sulak
U.S. Geological Survey

Malgorzata Konieczna
National Marine Fisheries Research Institute, Poland

See next page for additional authors

Characterization of Ichthyoplankton in the Northeastern Gulf of Mexico from Seamap Plankton Surveys, 1982-1999

Authors

Joanne Lyczkowski-Shultz, *National Oceanic and Atmospheric Administration*; David S. Hanisko, *National Oceanic and Atmospheric Administration*; Kenneth J. Sulak, *U.S. Geological Survey*; Malgorzata Konieczna, *National Marine Fisheries Research Institute, Poland*; and Pamela J. Bond, *National Oceanic and Atmospheric Administration*

CHARACTERIZATION OF ICHTHYOPLANKTON IN THE NORTHEASTERN GULF OF MEXICO FROM SEAMAP PLANKTON SURVEYS, 1982—1999

Joanne Lyczkowski–Shultz[1]*, David S. Hanisko[1], Kenneth J. Sulak[2], Małgorzata Konieczna[3], and Pamela J. Bond[1]

[1]*NOAA/NMFS/SEFSC Mississippi Laboratories, 3209 Frederic St., Pascagoula, MS 39568 USA; [2]U.S. Geological Survey, Southeast Ecological Science Center, 7920 NW 71st St., Gainesville, FL 32653 USA; [3]National Marine Fisheries Research Institute, Plankton Sorting and Identification Center, ul. Kazimierza Królewicza 4–E, 71–550 Szczecin, POLAND; *Corresponding author, email: Joanne.Lyczkowski–Shultz@noaa.gov

ABSTRACT: Data for 61 selected ichthyoplankton taxa from 1,166 bongo and neuston net samples at 72 stations comprising the USGS Northeastern Gulf of Mexico Ichthyoplankton Synopsis (UNIS) study area were analyzed. These data were taken during annual spring and fall Southeast Area Monitoring and Assessment Program (SEAMAP) Gulfwide plankton surveys over the period 1982–1999. The UNIS study area contributed disproportionately more fish eggs, total larvae and net–caught zooplankton biomass to survey totals than would be expected from the number of samples taken in the study area. This pattern was more evident during spring than fall surveys and is probably related to the close proximity of UNIS study area stations to the Mississippi River and the inshore penetration of nutrient rich deep slope water via the DeSoto Canyon. Statistical comparison of the percent frequency of occurrence of the 61 selected taxa revealed that the larvae of many were taken significantly more often in the UNIS study area than expected based on their occurrence Gulfwide. Thirteen of these taxa were taken more often in the study area during the season and collecting gear combination that accounted for the highest catches. These taxa represented fishes from mesopelagic, continental shelf, and reef assemblages reflecting the wide diversity of habitats available in the northeastern Gulf of Mexico. Distinct distribution patterns were observed among larvae in the UNIS study area that appear to be associated with the presence of the DeSoto Canyon. The consistent presence of fish eggs throughout the UNIS study area at mean abundances exceeding 100 eggs under 10 m² sea surface indicates that this region of the Gulf of Mexico is an important spawning area.

KEY WORDS: reef fishes, fish eggs, fish larvae, plankton displacement volume, DeSoto Canyon

INTRODUCTION

Hard–bottom and deep reef ecosystems in areas of hydrocarbon exploration and development in the northeastern Gulf of Mexico (NEGOM) have been the primary focus of integrated studies of fish communities of the outer continental shelf (OCS) by the U.S. Geological Survey (USGS)[1] under the Outer Continental Shelf Ecosystem Studies Program (Weaver et al. 1999, 2002, Sulak et al. 2000, Weaver and Sulak 2000, Gardner et al. 2001, Thurman et al. 2003). An ichthyoplankton component was added to these investigations in an attempt to address a fundamental deficiency in the knowledge of OCS ecosystems with the objective of assessing the composition, abundance, and geographic distribution patterns of fish eggs and larvae in the region. Of particular interest was acquisition of a baseline of knowledge on the larvae of fishes known to reside in OCS hard–bottom and deep–reef biotopes in order to better understand both zoogeographic and habitat factors determining demersal fish community structure and differentiation. Such baseline information may also prove valuable in assessing future anthropogenic impacts on the early life stages of fishes in areas of hydrocarbon exploration and development.

Previous studies of ichthyoplankton within the NEGOM region have detailed assemblage structure and seasonality, but were limited in duration (1–3 yr) and were conducted in adjoining but dissimilar habitats: Mississippi Sound and adjacent coastal waters (Rakocinski et al. 1996), Loop Current boundary in open Gulf of Mexico (GOM) waters (Richards et al. 1993), and the west Florida shelf southeast of Cape San Blas (Houde et al. 1979). The generalized description of seasonal occurrence, abundance and distribution of the early life stages of select taxa of fishes presented here is based on 1,166 ichthyoplankton samples collected during annual Southeast Area Monitoring and Assessment Program (SEAMAP; Rester et al. 2000) plankton surveys conducted by the National Marine Fisheries Service over the period 1982–1999. The objective of our study was to analyze SEAMAP ichthyoplankton data from 18 years of plankton surveys and, although not designed to elucidate biological/physical coupling and recruitment dynamics, the results presented here depict 'average conditions' for selected taxa in continental shelf and offshore waters of the NEGOM between the Mississippi River and Cape San Blas, Florida.

MATERIALS AND METHODS

SEAMAP Surveys and Collections

The SEAMAP sampling area encompasses the northern GOM from the 10 m isobath out to the U.S. Exclusive Economic Zone (EEZ). Although about 300 sampling sites were initially proposed Gulfwide, only about 200 stations have been consistently targeted for sampling during SEAMAP

[1]*Southeast Ecological Science Center, Coastal Ecology and Conservation (CEC) Research Group, Gainesville, Florida.*
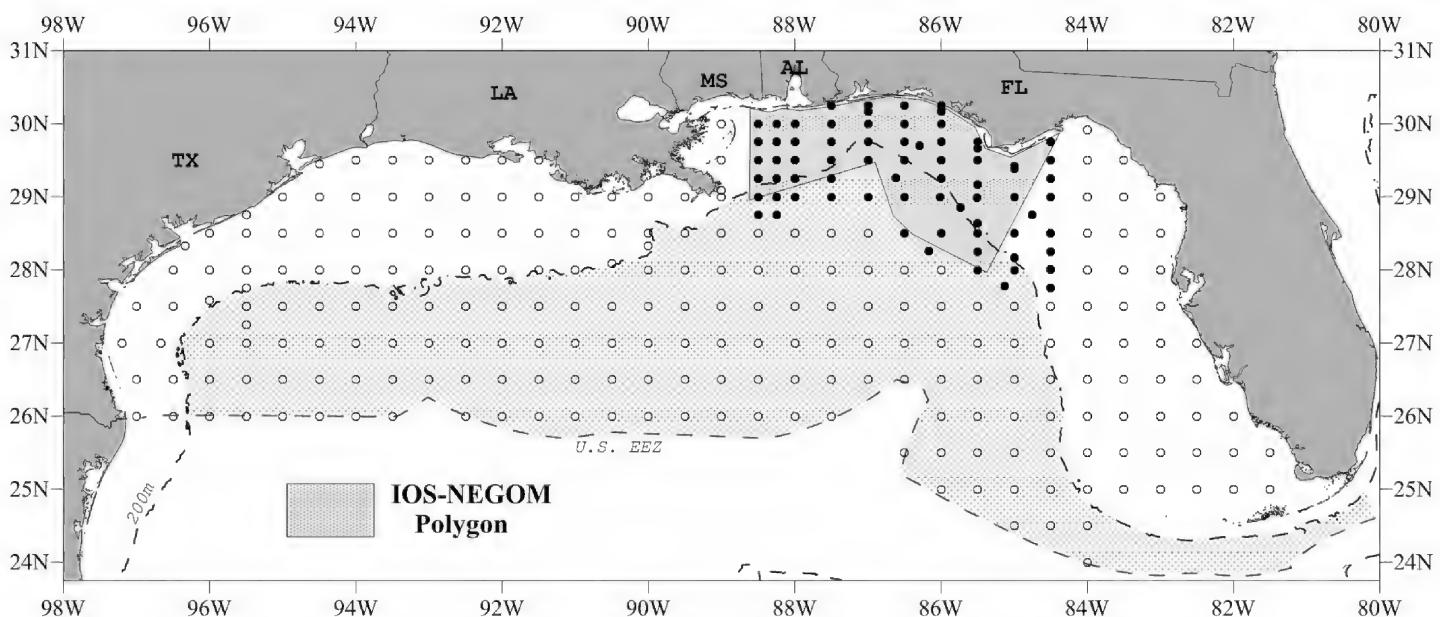
Figure 1. *Location of Southeast Area Monitoring and Assessment Program (SEAMAP) ichthyoplankton stations in the Gulf of Mexico, 1982-1999 surveys (all circles). During fall, survey stations were generally located inside the 200 m isobath; during spring, survey stations were located outside the 200 m isobath (lighter shaded area). Survey stations located within the USGS Northeastern Gulf of Mexico Ichthyoplankton Synopsis (UNIS) study area (filled circles) extend outside of the Integrated Oceanographic Study - Northeastern Gulf of Mexico (IOS-NEGOM) polygon (darker shaded area). UNIS stations were sampled during fall and/or spring SEAMAP surveys. U.S. EEZ = United States Exclusive Economic Zone boundary.*

surveys (Figure 1). A subset of 72 SEAMAP stations (fixed geographic locations) lying in and adjacent to the USGS study area or IOS–NEGOM (Integrated Oceanographic Study – northeastern Gulf of Mexico) research polygon was selected for analysis. This subset of stations identified by SEAMAP station number, and referred to hereafter as the UNIS (USGS NEGOM Ichthyoplankton Synopsis) study area is bounded to the east by longitude 84.5°W, and to the west by longitude 88.5°W. It extends from the 10 m isobath seaward to about the 1,000 m isobath (Figure 2).

Although SEAMAP plankton sampling in the GOM has been consistently conducted during 4 survey time frames since 1982 (Lyczkowski–Shultz and Hanisko 2007), only surveys conducted in spring/early summer and late summer/early fall months covered the entire extent of the UNIS study area. Therefore, this characterization of ichthyoplankton in the UNIS study area is based on a subset of data from 32 SEAMAP plankton surveys conducted during the period 1982–1999 (Lyczkowski–Shultz et al. 2004). Two comparative areas and seasons were sampled: 1) the open GOM during April, May and June, 1982–1984 and 1986–1999, referred to hereafter as, the 'spring' survey, and 2) the continental shelf typically out to 200 m in August 1984 and September to early October, 1986–1999, referred to hereafter as the 'fall' survey.

The sampling gear and methodology used during SEAMAP surveys are similar to those recommended by

Kramer et al. (1972), Smith and Richardson (1977) and Posgay and Marak (1980). A 61 cm bongo net fitted with 0.335 mm mesh netting was fished in an oblique tow path to a maximum depth of 200 m or to 2–5 m off the bottom at depths < 200 m. A mechanical flowmeter was mounted off–center in the mouth of each bongo net to record the volume of water filtered. Volumes filtered ranged from 22–555 m³ but were typically 30–40 m³ at the shallowest stations and 300–400 m³ at the deepest stations in the UNIS study area. A single or double 2 x 1 m pipe frame neuston net fitted with 0.950 mm mesh netting was towed at the surface with the frame half–submerged for 10 min. The neuston net was not fitted with a flowmeter; therefore, filtered water volume was not measured for neuston samples. Samples were taken upon arrival on station regardless of time of day. The number of samples collected in daytime and nighttime hours was about equal during both seasons surveyed over the time period represented by this study.

Environmental data consistently gathered during SEAMAP surveys include salinity (psu), temperature (°C), dissolved oxygen (mg/L), and since ca. 1993 optical transmission (%) and fluorescence (μg/l) (see Rester et al. 2000 for complete description). Although not presented here, SEAMAP environmental data are available upon request from the SEAMAP Data Manager[2].

Most SEAMAP stations were located at 30 nautical mile or 0.5° (~56 km) intervals in a fixed, systematic grid

of transects across the GOM. Occasionally during surveys, samples were moved to avoid navigational hazards or were taken at non–standard locations or stations. This was especially true during Oregon II cruise 146 in August 1984 when additional samples were taken at locations between standard SEAMAP stations. These stations are denoted with the prefix 'U' as opposed to the prefix 'B' or 'A' for standard SEAMAP stations in Figure 2. Data from these stations were also included in the analysis. At each station either a bongo and/or neuston tow was conducted depending on the specif-
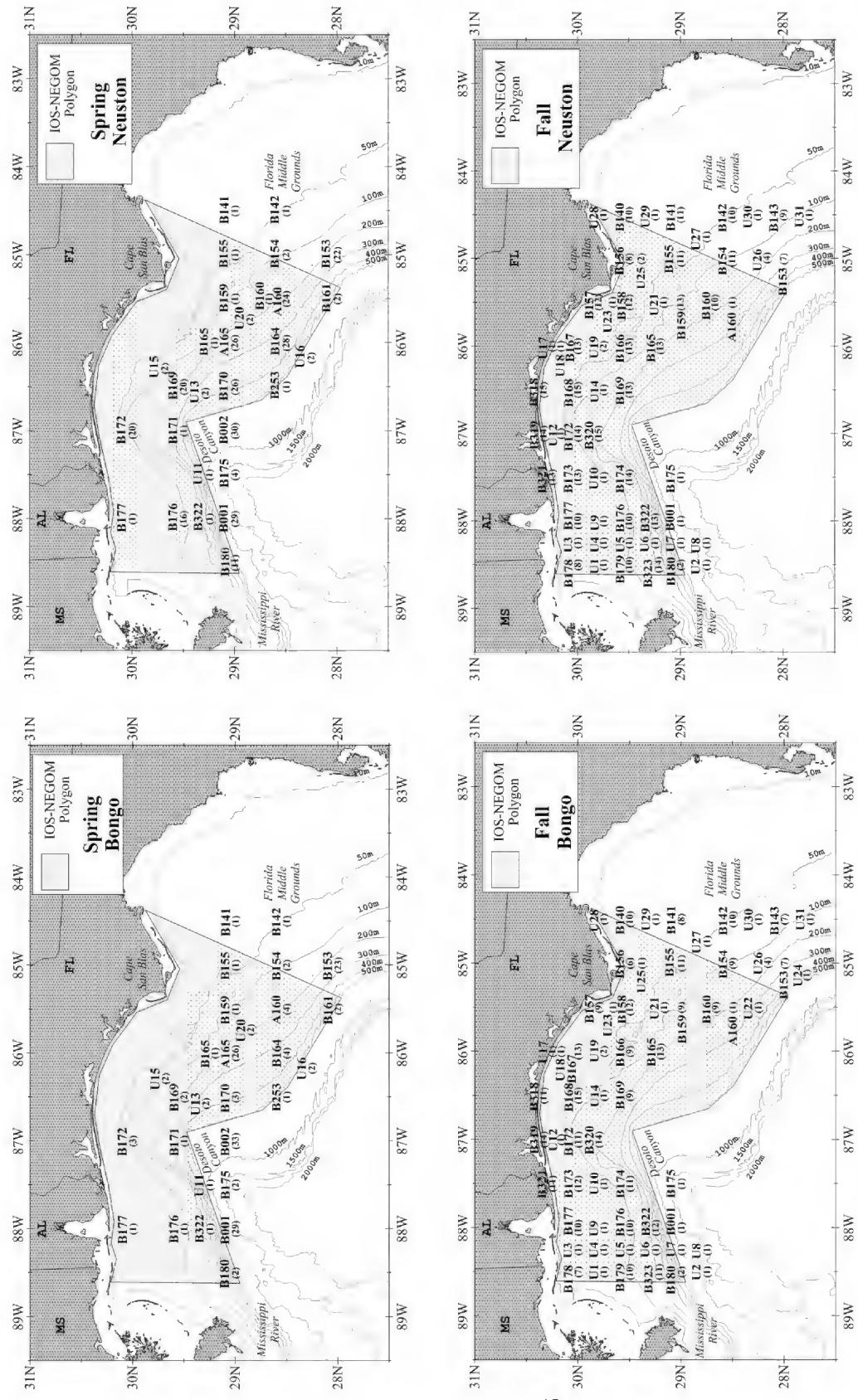
Figure 2. Location of the 72 stations comprising the UNIS Study area, including the smaller Integrated Oceanographic Study - Northeastern Gulf of Mexico (IOS-NEGOM) polygon (shaded), where ichthyoplankton samples were collected during SEAMAP plankton surveys over the period 1982-1999. Station identifier and the number of samples collected (in parenthesis) at that station are shown for each of the 2 seasons sampled (spring and fall) and each of the 2 plankton gear types (bongo and neuston) used during plankton surveys. UNIS and SEAMAP defined in Figure 1.

ic survey. During spring surveys, bongo tows were made only at every other station and the targeted survey stations were sampled twice, i.e. 2 transits over the survey station grid were completed. However, only a single transit over the targeted survey area was completed during fall surveys. This accounts for the differences in total number of samples collected over the time series at stations in the study area during spring and fall surveys (Figure 2).

Of the 72 stations representing the UNIS study area, 12 were sampled only during spring surveys, 43 only during fall surveys, and 17 during both survey types. There was 35–40% overlap in spatial coverage during spring and fall surveys (Figure 2). Most spring survey stations were located seaward of the 50 m isobath, whereas more than half of fall stations were located shoreward of the 50 m isobath. A detailed listing of the SEAMAP surveys, sampling dates, station positions and station depths that provided ichthyoplankton data presented here can be found in Lyczkowski–Shultz et al. (2004; http://fl.biology.usgs.gov/coastaleco).

Sample Processing and Ichthyoplankton Identifications

Initial processing of SEAMAP plankton samples was accomplished at the National Marine Fisheries Research Institute, Plankton Sorting and Identification Center, in Szczecin, Poland, under a Joint Studies Agreement with NMFS. Wet plankton volumes of bongo net samples were measured by displacement to estimate net–caught zooplankton biomass (Smith and Richardson 1977). Fish eggs and larvae were removed from bongo net samples, and fish larvae only from neuston net samples. Fish egg counts were not quantitative for some samples during the early years of the SEAMAP time series. These samples were not used in calculations of mean egg abundance and this accounts for differences in the number of samples used to calculate mean egg and larval abundances that are presented here. Larvae were identified to the lowest possible taxon (to family in most cases). Body length (BL) in mm (either notochord or standard length) was measured for a varying number (2 to all specimens) depending on the taxonomic level of identification. Typically, all or up to 10 specimens were measured for larvae identified to species and in some instances genus. Only size range (i.e., size of the largest and smallest specimens) was recorded for larvae identified to family and higher levels. Mean length and/or length range are presented here as appropriate for the taxonomic level of identification. However, both mean length and range are summarized for 16 taxa of reef fishes regardless of taxonomic level of identification. For those 16 taxa, mean length was based only on samples where all captured specimens were measured. Vials of eggs and identified larvae, plankton displacement volume values, total egg counts, and counts and length measurements of identified larvae were sent to the SEAMAP Archiving Center (SAC) at the Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute (formerly the Florida

Marine Research Institute), St. Petersburg, FL. All data have been entered into the SEAMAP database that is maintained at the NMFS Mississippi Laboratories, Pascagoula Facility in Pascagoula, MS. Voucher specimens are curated at SAC, and are available on loan for scientific study and reference.

The majority of specimens collected in SEAMAP plankton samples and maintained at the Archive have been identified only to the family level. This is not unexpected since up until the recent publication of an identification guide to the early life stages of fishes in the western central north Atlantic (Richards 2006), the larvae of only 27% of the ~1,800 species of marine fishes occurring in the western central north Atlantic region (including the GOM) had been described (Kendall and Matarese 1994). Identification of larvae to the family level, however, is possible for over two–thirds of the families of marine fishes (Ahlstrom and Moser 1981; Richards 1990, 2006).

We summarized data for only a limited number of taxa due to the limitations of larval fish identifications in the taxonomically rich GOM. Moreover, the large number of specimens available from SEAMAP surveys (>100,000 at the time this analysis was undertaken) made it impractical to re–examine specimens using newly available descriptive information. Therefore, only larvae of 61 taxa, representing 34 families of fishes, were chosen for analysis because their larvae are distinctive and can be identified with confidence to family, subfamily, genus, or species (Table 1). Also, in the case of the tunas (from spring survey samples), mackerels and snappers (both spring and fall samples), most specimens have been re–examined and, as necessary, re–identified by NMFS ichthyoplankton experts prior to use of larval abundance data in stock assessments.

Taxa selected for treatment herein were chosen using these criteria: 1) larvae could be reliably identified throughout the time series; 2) larvae had been re–examined to validate identifications; and 3) larvae were identified as belonging to selected families considered as being consistently associated with reef environments (Sale 1991). Although identification of larvae in these latter families [Holocentridae, Serranidae (in part), Priacanthidae, Apogonidae, Haemulidae, Chaetodontidae, Pomacanthidae, Pomacentridae, Labridae, Scaridae, and Acanthuridae] remains problematic, adults in these taxa often comprise key members of OCS hard–bottom and deep–reef communities in the NEGOM.

This subset of taxa chosen for analysis of ichthyoplankton in the UNIS study area represents the wide diversity of the NEGOM ichthyofauna and includes both key ecological and resource species. These taxa are representative of the tropical and warm temperate epipelagic, mesopelagic, coastal shelf and demersal (including reef), and pelagic species found in the northern GOM (Richards et al. 1993).

Data Summaries and Comparisons

Catches of total fish eggs, total fish larvae and larvae of

selected taxa in bongo net samples were standardized to account for sampling effort and expressed as number under 10 m^2 sea surface by dividing the number of eggs or larvae by volume filtered and then multiplying the resultant by the product of 10 and maximum depth of tow. This standardization results in a less biased estimate of abundance than number per unit of volume filtered alone and permits direct comparison of abundance estimates across samples taken over a wide range of water column depths (Smith and Rich-

ardson 1977). Plankton displacement volumes from bongo nets were standardized using the same methodology as for fish eggs and larvae but are expressed as cc per 10 m^2 sea surface. Standardized catches or catch per unit effort (CPUE) of total fish larvae and larvae of selected taxa taken in neuston samples were expressed as number/10 min tow. Standardized catches of total fish larvae include all taxa taken in a sample and not just those selected for detailed analysis.

Mean values at UNIS stations for bongo and neuston

TABLE 1. *Catch data for 61 selected fish taxa analyzed from the USGS Northeastern Gulf of Mexico Ichthyoplankton Synopsis (UNIS) study area. Data include number of occurrences (Occ.) and the number of specimens collected in bongo and neuston samples during Southeast Area Monitoring and Assessment Program (SEAMAP) spring and fall plankton surveys over the period 1982–1999. Reef–associated species and higher taxa that include characteristically reef–associated species are denoted in bold. n = number of samples*

| | | | Sampling Gear | | | | Survey | | | |
| | | | Bongo n = 499 | | Neuston n = 667 | | Spring n = 433 | | Fall n = 733 | |
Taxon	Total Occ.	Total Number Specimens	Occ.	Number Specimens	Occ.	Number Specimens	Occ.	Number Specimens	Occ.	Number Specimens
Elopidae	7	16	3	5	4	11	0	0	7	16
Muraenidae	83	188	23	29	60	159	12	21	71	167
Engraulidae	573	40732	283	15345	290	25387	97	3100	476	37632
Clupeidae										
Etrumeus teres	56	1306	31	354	25	952	53	1303	3	3
Harengula jaguana	137	3909	30	140	107	3769	47	2440	90	1469
Opisthonema oglinum	89	1163	50	794	39	369	10	223	79	940
Sardinella aurita	148	4360	67	1616	81	2744	8	35	140	4325
Sternoptychidae	210	3533	204	3411	6	122	122	2310	88	1223
Synodontidae	501	7229	360	6497	141	732	140	1253	361	5976
Paralepididae	215	1053	196	1028	19	25	130	775	85	278
Carapidae	22	62	21	61	1	1	5	6	17	56
Carapus bermudensis	68	210	38	210	0	0	8	11	60	199
Bregmacerotidae	441	9918	371	9474	70	444	182	2933	259	6985
Mugilidae	154	1669	15	36	139	1633	144	1647	10	22
Melamphaidae	58	90	51	82	7	8	41	66	17	24
Holocentridae	23	34	5	5	18	29	10	17	13	17
Serranidae	320	1415	263	1272	57	143	88	317	232	1098
Serraninae	236	1672	150	1118	86	554	54	235	182	1437
Anthiinae	72	182	56	126	16	56	43	112	29	70
Epinephelinae	3	3	3	3	0	0	3	5	0	0
Grammistinae	117	215	78	143	39	72	17	41	100	174
Priacanthidae	109	239	55	102	54	137	18	57	91	182
Apogonidae	169	579	98	342	71	237	58	175	111	404
Rachycentridae										
Rachycentron canadum	5	21	0	0	5	21	3	17	2	4
Coryphaenidae	187	438	27	37	160	401	109	281	78	157
Carangidae										
Caranx spp.	183	1449	37	91	146	1358	90	812	93	637
Chloroscombrus chyrsurus	206	14916	98	1217	108	13699	1	8	205	2908
Decapterus spp.	479	7101	226	3832	253	3269	60	397	419	6704
Selar crumenophthalmus	99	710	54	122	45	588	4	4	95	706
Selene spp.	34	53	17	28	17	25	0	0	34	53
***Seriola* spp.**	123	461	8	12	115	449	80	385	43	76
Trachinotus spp.	46	85	0	0	46	85	32	60	14	25
Trachurus lathami	16	61	6	17	10	44	16	61	0	0
Lutjanidae	190	728	162	668	28	60	8	8	182	720
Lutjanus spp.	34	64	18	23	16	41	0	0	34	64

TABLE 1. Continued

Taxon	Total Occ.	Total Number Specimens	Sampling Gear				Survey			
			Bongo n = 499		Neuston n = 667		Spring n = 433		Fall n = 733	
			Occ.	Number Specimens	Occ.	Number Specimens	Occ.	Number Specimens	Occ.	Number Specimens
Lutjanus campechanus	33	71	10	14	23	57	1	2	32	69
Lutjanus griseus	9	9	6	6	3	3	1	1	8	8
Pristipomoides aquilonaris	74	208	51	124	23	84	5	6	69	202
Rhomboplites aurorubens	174	644	114	318	60	326	4	21	170	623
Lobotidae										
Lobotes surinamensis	23	39	0	0	23	39	2	2	21	37
Haemulidae	10	139	7	136	3	3	2	2	8	137
Sciaenidae										
Cynoscion spp.	64	515	42	412	22	103	4	19	60	496
Sciaenops ocellata	48	351	32	243	16	108	0	0	48	351
Mullidae	268	19855	31	91	237	19764	241	19651	27	204
Chaetodontidae	11	12	4	5	7	7	2	2	9	10
Pomacanthidae	6	7	2	3	4	4	4	4	2	3
Pomacentridae	63	166	33	90	30	76	16	82	47	84
Labridae	358	3420	288	3180	70	240	50	165	308	3255
Scaridae	113	369	90	331	23	38	28	39	85	330
Acanthuridae	4	5	2	2	2	3	3	4	1	1
Trichiuridae										
Trichiurus lepturus	82	260	69	222	13	38	16	62	66	198
Scombridae										
Acanthocybium solandri	2	2	2	2	0	0	0	0	2	2
Katsuwonus pelamis	63	136	36	60	27	76	34	66	29	70
Scomberomorus cavalla	87	143	44	55	43	88	0	0	87	143
Scomberomorus maculatus	39	144	19	39	20	105	12	85	27	59
Thunnus spp.	165	712	76	186	89	526	45	209	120	503
Thunnus thynnus	26	136	7	13	19	123	26	136	0	0
Xiphiidae										
Xiphias gladius	3	4	0	0	3	4	3	4	0	0
Istiophoridae	38	78	4	7	34	71	13	27	25	51
Stromateidae										
Peprilus alepidotus	51	181	36	136	15	45	1	1	50	180
Peprilus burti	115	813	92	721	23	92	8	20	107	793

nets by survey (spring and fall plankton) were based on all samples collected at each sampling station during surveys over the period 1982–1999. Mean values of the aforementioned data categories (total eggs, total fish larvae, plankton sample displacement volume, and taxon–specific standardized catches) are represented on distribution maps (Figures 3–62). Mean abundance values for each selected taxon by area, season and gear combination are not directly reported here, but, these values can be calculated from the information found in Tables 2 and 4[3].

The percent frequency of occurrence for each taxon, gear and season combination was tested to determine if the UNIS study area value was significantly higher or lower than the expected value from Gulfwide sampling. Comparisons were carried out using a one sample binomial test for proportions (alpha = 0.05) utilizing the FREQ procedure in SAS (Version 9.3 of the SAS System for Windows). All comparisons were one sided based on whether the difference between the study area and Gulfwide values were less than or greater than zero. Comparisons were only carried out for combinations in which a taxon occurred in both the study area and Gulfwide samples. All p–values for percent frequency are presented in Table 3.

Direct comparison of ichthyoplankton abundance in the UNIS study area to the entire survey area Gulfwide is problematic due the zero–inflated nature of ichthyoplankton

---

[3]Mean abundance or CPUE is equal to the percent of total abundance (PTA) or percent of total CPUE (PTC) from Table 4 divided by 100 then multiplied by the total standardized abundance or CPUE in Table 2 and divided by the number of samples in Table 4 for the appropriate area, season and gear combination. Due to rounding of data presented in the tables, the calculated values will be approximate but very close to mean values calculated from the actual samples.

data, and the large difference between the 2 areas in number of samples collected (1,166 vs. 7,100, respectively). In order to overcome this difficulty, comparisons were made using metrics based on the percent of total standardized abundance or CPUE. Comparisons between the UNIS study area and Gulfwide sampling for fish eggs, total fish larvae (all taxa
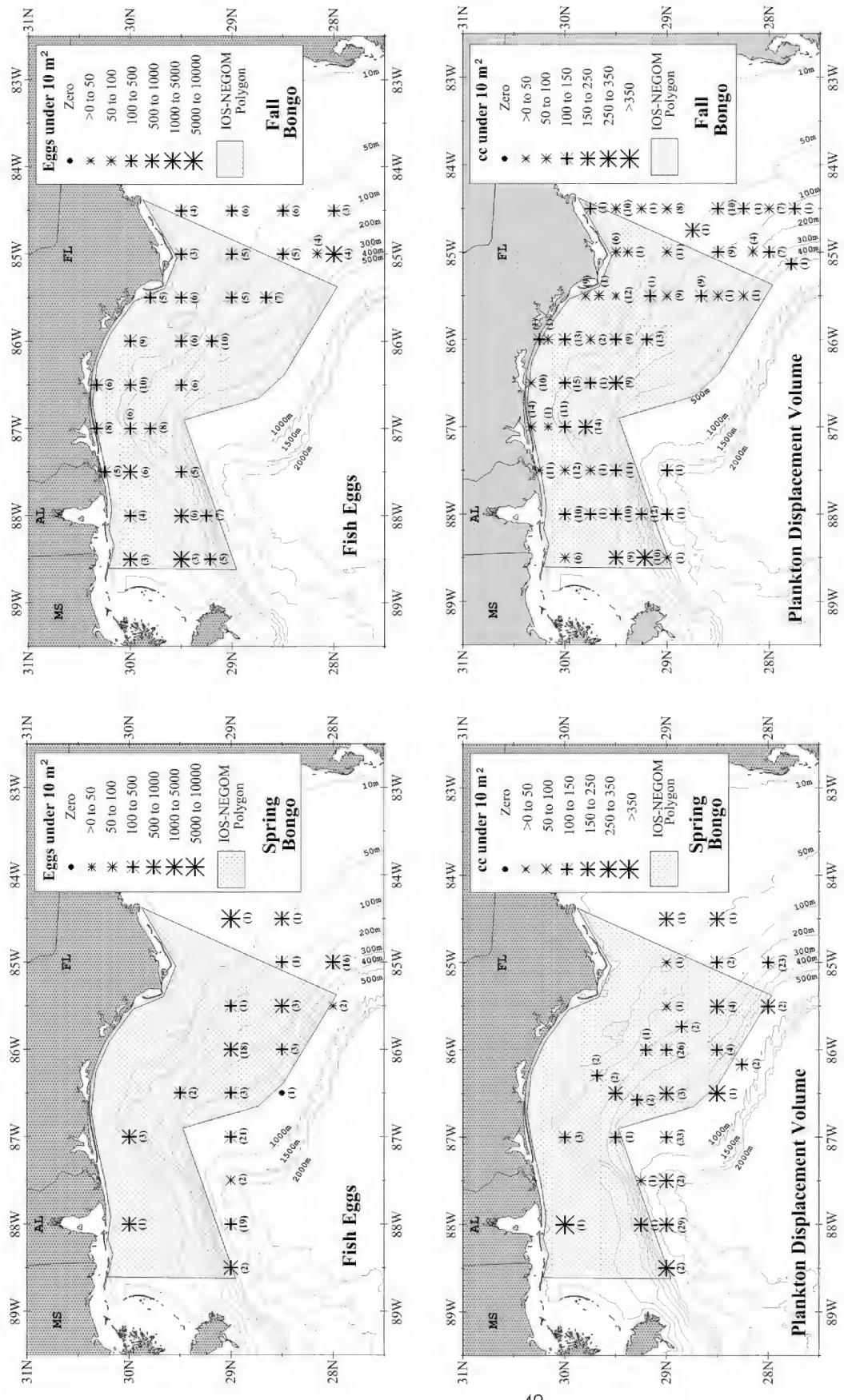

Figure 3. Mean abundance of fish eggs and mean plankton displacement volumes at UNIS study area stations during SEAMAP spring and fall plankton surveys, 1982-1999. Number of bongo samples used to calculate mean values is in parenthesis. UNIS and SEAMAP defined in Figure 1.

TABLE 2. *The number of USGS Northeastern Gulf of Mexico Ichthyoplankton Synopsis (UNIS) study area samples, total standardized abundance or CPUE of fish eggs and total fish larvae, and total standardized plankton displacement volumes expressed as a percent of the corresponding totals from Gulfwide sampling during SEAMAP spring and fall plankton surveys over the period, 1982-1999.*

	Number of Samples			Total Standardized Abundance or CPUE		
	UNIS	Gulfwide	% UNIS/Gulfwide	UNIS	Gulfwide	% UNIS/Gulfwide
Fish Eggs						
Spring Bongo	100	939	11	46,658.50	225,138.8	21
Fall Bongo	176	836	21	71,708.20	354,986.8	20
Total Larvae						
Spring Bongo	154	1,453	11	208,493.91	1,328,800.6	16
Spring Neuston	279	2,290	12	41,242.82	194,334.1	21
Fall Bongo	345	1,591	22	399,675.54	199,8674.1	20
Fall Neuston	388	1,766	22	56,908.48	356,974.1	16
Displacement Volume						
Spring Bongo	153	1,449	11	22,803.16	180,899.6	13
Fall Bongo	332	1,549	21	36,962.87	130,575.6	28

combined) and total standardized plankton displacement volumes were made based on the ratio of the UNIS study area total divided by the Gulfwide total and then multiplied by 100 (% UNIS/Gulfwide). Comparisons of the relative abundances of the selected taxa were made using the percent of total abundance (PTA) for bongo samples, and percent of total CPUE (PTC) for neuston samples. PTA and PTC were calculated by taking the total standardized abundance of an individual taxon and dividing it by the total standardized abundance of total fish larvae and multiplying by 100. The PTA/PTC metrics were calculated for each study area, season and gear combination.

Results and Discussion

Survey Summary Information

Over 7,000 plankton samples were taken during 17 spring (April, May and June) and 15 fall (August, September to early October) SEAMAP Gulfwide plankton surveys. A subset of 1,166 of these samples (499 bongo and 667 neuston samples) from these surveys were used to characterize ichthyoplankton in the UNIS study area (Figure 1, Table 2).

No attempt was made to identify fish eggs from SEAMAP samples as identification to even the family level is exceedingly difficult especially in bodies of water with high diversity such as the GOM. Fish eggs were present in bongo samples throughout the UNIS study area (Figure 3). Mean egg abundances at stations where the number of samples (n) sorted for eggs was > 1 generally ranged between 120–600 in the spring and 200–400 in the fall. Mean (± se) abundance from spring surveys was 467 ± 76 (n = 100), and from fall surveys 407 ± 35 (n = 176) eggs/10 m². The number of samples sorted for fish eggs in the UNIS study area represented 11% of the total number of samples with egg counts for all spring survey samples, yet the total standardized abundance of fish eggs in the UNIS study area accounted for 21% of the Gulfwide total

(Table 2). Total standardized abundance of fish eggs in the UNIS study area during fall surveys reflected the proportion of total survey samples taken in the area, about 20%.

Mean plankton displacement volume for all stations in the UNIS study area combined was higher in spring, 149 ± 7 (n = 153) than in fall, 111 ± 4 (n = 332) cc/10 m² (Figure 3). Mean displacement volumes > 150 cc/10 m² in spring and > 100 cc/10 m² in fall were observed throughout the study area. During both spring and fall surveys, the contribution of UNIS study area samples to the sum of standardized plankton displacement volumes Gulfwide was proportionately higher than would be expected based on the allocation of samples in the area (Table 2). During spring surveys, the study area accounted for only 11% of the total number of survey samples but the total standardized plankton displacement volumes of these samples accounted for 13% of the entire spring survey total. During fall surveys, the study area contributed 21% of the total number of survey samples but total standardized plankton displacement volume of these samples accounted for 28% of the entire fall survey total.

Fish larvae were taken in each of the 499 bongo net collections and in all but 2 of the 667 neuston net collections in the UNIS study area. Overall mean abundance of total fish larvae (all taxa combined) from the two seasons were similar; 1354 ± 80 (n = 154) and 1158 ± 54 (n = 345) larvae under 10 m², and 148 ± 33 (n = 279) and 147 ± 19 (n = 388) larvae/10 min tow, in spring and fall surveys, respectively. Mean abundances of total fish larvae in bongo net samples at stations where the number of samples (n) was > 1 ranged from 529–2745 and 302–2239 larvae under 10 m² during spring and fall surveys, respectively. Mean abundances of total fish larvae in neuston collections at stations where n > 1 ranged from 43–571 and 27–1140 larvae/10 min tow in spring and fall surveys, respectively. Mean abundances of total fish larvae were relatively uniform throughout the

area especially where estimates were based on more than 5 samples (Figure 4).

The relative contribution of UNIS study area samples to the overall standardized abundance of total fish larvae differed somewhat between the 2 seasonal surveys (Table 2). In spring, the total catch of all fish larvae captured by bongo
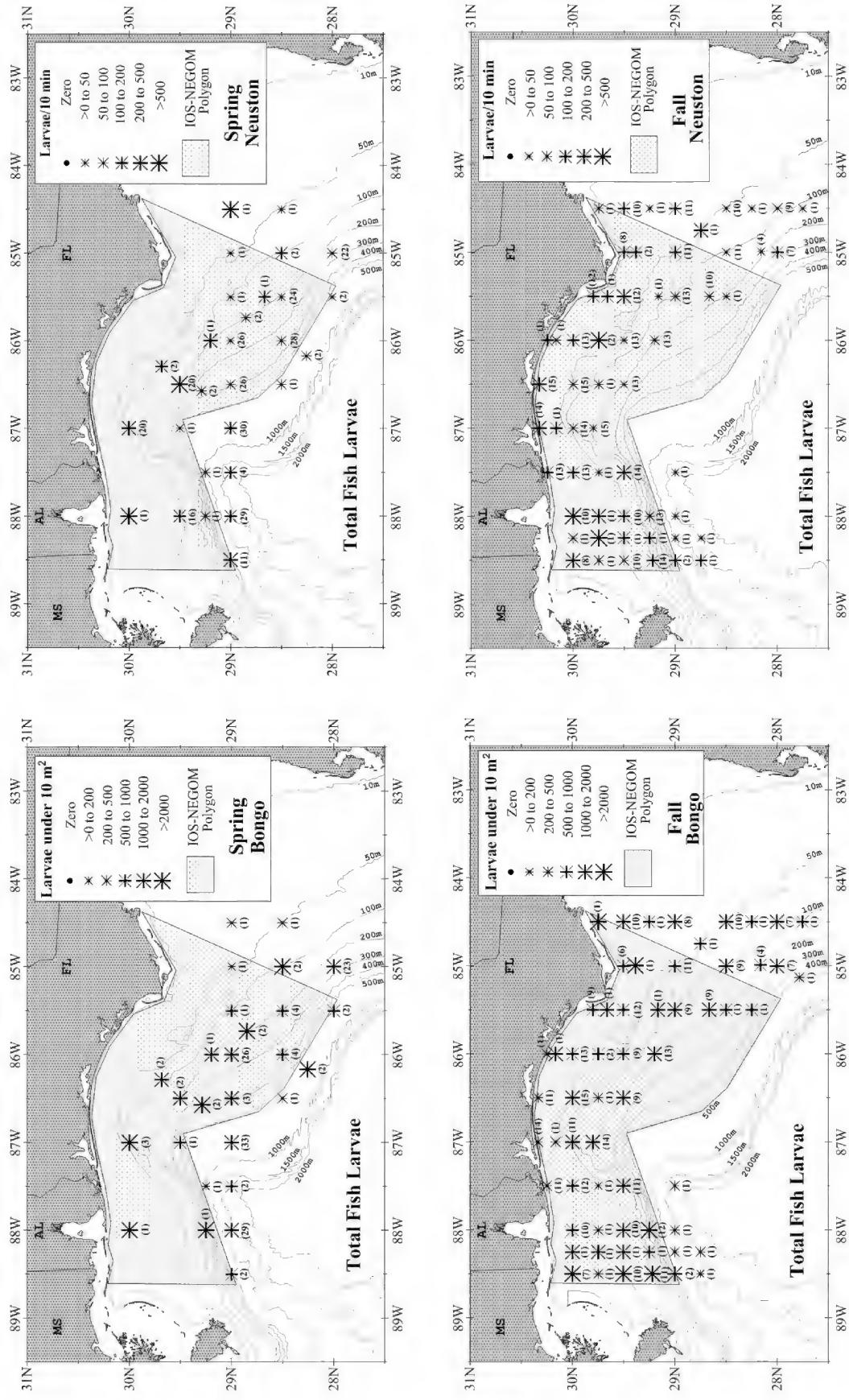

Figure 4. Mean abundance and mean CPUE of total fish larvae (all taxa combined) collected in spring and fall, bongo and neuston samples at UNIS study area stations during SEAMAP plankton surveys over the period 1982-1999. Number of samples used to calculate mean values is indicated in parenthesis. UNIS and SEAMAP defined in Figure 1.

TABLE 3. *Summary of a one sided, binomial test (α = 0.05) for proportions on the percent frequency of occurrence of 61 select ichthyoplankton taxa captured in USGS Northeastern Gulf of Mexico Ichthyoplankton Synopsis (UNIS) study area samples alone and in all Gulfwide (GOM) survey samples combined during SEAMAP spring and fall plankton surveys over the period 1982-1999. FO = frequency of occurrence. Diff. = UNIS %FO minus GOM %FO. n = number of samples. (p) = p—valve. p—values in bold indicate a statistically significant difference.*

	Spring Bongo			Spring Neuston			Fall Bongo			Fall Neuston		
Taxon	UNIS %FO n = 154	GOM %FO n = 1453	Diff. (p)	UNIS %FO n = 279	GOM %FO n = 2290	Diff. (p)	UNIS %FO n = 345	GOM %FO n = 1591	Diff. (p)	UNIS %FO n = 388	GOM %FO n = 1766	Diff. (p)
Elopidae							0.87	0.82	0.05 (0.457)	1.03	0.79	0.24 (0.298)
Muraenidae	1.30	2.48	-1.18 (0.173)	3.58	3.54	0.05 (0.483)	6.09	6.47	-0.39 (0.385)	12.89	9.12	3.77 **(0.005)**
Engraulidae	23.38	13.15	10.23 **(<0.001)**	21.86	10.61	11.25 **(<0.001)**	71.59	62.10	9.49 **(<0.001)**	59.02	48.64	10.38 **(<0.001)**
Etrumeus teres	18.83	8.12	10.71 **(<0.001)**	8.60	2.27	6.33 **(<0.001)**	0.58	0.44	0.14 (0.348)	0.26	0.17	0.09 (0.337)
Harengula jaguana	3.90	1.79	2.11 **(0.024)**	14.70	5.94	8.76 **(<0.001)**	6.96	14.02	-7.06 **(<0.001)**	17.01	25.03	-8.02 **(<0.001)**
Opisthonema oglinum	3.25	1.03	2.21 **(0.003)**	1.79	0.52	1.27 **(0.002)**	13.04	31.93	-18.89 **(<0.001)**	8.76	22.76	-14.00 **(<0.001)**
Sardinella aurita	0.65	1.72	-1.07 (0.153)	2.51	2.71	-0.20 (0.419)	19.13	21.62	-2.49 (0.131)	19.07	20.95	-1.88 (0.182)
Sternoptychidae	75.97	68.27	7.70 **(0.020)**	1.79	0.96	0.83 (0.077)	25.22	18.23	6.99 **(<0.001)**	0.26	0.34	-0.08 (0.391)
Synodontidae	53.90	35.10	18.80 **(<0.001)**	20.43	10.17	10.26 **(<0.001)**	80.29	66.00	14.29 **(<0.001)**	21.65	19.03	2.62 (0.094)
Paralepididae	74.68	70.41	4.27 (0.123)	5.38	5.24	0.14 (0.459)	23.48	18.79	4.69 **(0.013)**	1.03	0.45	0.58 **(0.045)**
Carapidae	3.25	1.65	1.59 (0.060)				4.64	2.51	2.12 **(0.006)**	0.26	0.06	0.20 **(0.048)**
Carapus bermudensis	5.19	4.34	0.86 **(<0.001)**				17.39	9.87	7.52 (0.300)			
Bregmacerotidae	92.21	80.18	12.03 **(<0.001)**	14.34	6.51	7.83 **(<0.001)**	66.38	58.20	8.17 **(<0.001)**	7.73	5.95	1.79 (0.068)
Mugilidae	8.44	5.44	3.00 (0.050)	46.95	30.70	16.25 **(<0.001)**	0.58	2.51	-1.93 **(0.011)**	2.06	13.36	-11.30 **(<0.001)**
Melamphaidae	23.38	33.10	-9.73 **(0.005)**	1.79	0.96	0.83 (0.077)	4.35	5.47	-1.12 **(0.180)**	0.52	0.28	0.23 (0.195)
Holocentridae	0.65	4.82	-4.17 **(0.008)**	3.23	7.64	-4.42 **(0.003)**	1.16	3.33	-2.17 **(0.012)**	2.32	4.02	-1.70 **(0.044)**
Serranidae	39.61	31.11	8.50 **(0.011)**	9.68	9.00	0.68 (0.345)	58.55	48.90	9.65 **(<0.001)**	7.73	6.96	0.77 (0.276)
Serraninae	14.29	17.00	-2.71 (0.185)	11.47	7.03	4.44 **(0.002)**	37.10	35.26	1.84 (0.237)	13.92	12.74	1.18 (0.243)
Anthiinae	19.48	18.51	0.97 (0.379)	4.66	3.89	0.77 (0.252)	7.54	4.34	3.20 **(0.002)**	0.77	0.40	0.38 (0.119)
Epinephelinae	1.95	1.86	0.09 (0.467)									
Grammistinae	4.55	4.68	-0.13 (0.469)	3.58	2.79	0.79 (0.212)	20.58	17.60	2.98 (0.073)	7.47	7.42	0.06 (0.483)
Priacanthidae	3.90	5.37	-1.47 (0.209)	4.30	4.24	0.07 (0.478)	14.20	14.20	0.00 (0.500)	10.82	10.02	0.80 (0.299)
Apogonidae	14.94	13.76	1.17 (0.337)	12.54	7.16	5.38 **(<0.001)**	21.74	28.66	-6.92 **(0.002)**	9.28	11.66	-2.39 (0.072)
Rachycentron canadum				1.08	0.79	0.29 (0.292)				0.52	3.11	-2.60 **(0.002)**

TABLE 3. *Continued*

Taxon	Spring Bongo			Spring Neuston			Fall Bongo			Fall Neuston		
	UNIS %FO n = 154	GOM %FO n = 1453	Diff. (p)	UNIS %FO n = 279	GOM %FO n = 2290	Diff. (p)	UNIS %FO n = 345	GOM %FO n = 1591	Diff. (p)	UNIS %FO n = 388	GOM %FO n = 1766	Diff. (p)
Coryphaenidae	9.74	12.11	-2.37 (0.183)	33.69	39.34	-5.65 **(0.027)**	3.48	2.89	0.59 (0.258)	17.01	14.95	2.06 (0.127)
Caranx spp.	12.34	17.14	-4.8 (0.057)	25.45	39.56	-14.12 **(<0.001)**	5.22	9.87	-4.65 **(0.002)**	19.33	24.75	-5.42 **(0.007)**
Chloroscombrus chyrsurus				0.36	0.57	-0.21 (0.321)	28.41	45.00	-16.60 **(<0.001)**	27.58	44.11	-16.53 **(<0.001)**
Decapterus spp.	7.14	7.43	-0.29 (0.445)	17.56	12.10	5.47 **(0.003)**	62.32	38.91	23.41 **(<0.001)**	52.58	31.77	20.81 **(<0.001)**
Selar crumenophthalmus	1.30	2.55	-1.25 (0.163)	0.72	3.67	-2.95 **(0.004)**	15.07	16.28	-1.21 (0.272)	10.82	11.72	-0.90 (0.292)
Selene spp.							4.93	11.00	-6.07 **(<0.001)**	4.38	5.44	-1.05 (0.180)
Seriola spp.	3.25	3.58	-0.33 (0.412)	26.88	20.83	6.05 **(0.006)**	0.87	2.33	-1.46 **(0.036)**	10.31	10.31	0.00 (0.499)
Trachinotus spp.				11.47	8.47	3.00 **(0.036)**				3.61	5.27	-1.66 (0.072)
Trachurus lathami	3.90	2.34	1.56 (0.101)	3.58	2.79	0.79 (0.212)						
Lutjanidae	3.25	4.13	-0.88 (0.291)	1.08	0.96	0.11 (0.422)	45.51	42.43	3.08 (0.123)	6.44	6.29	0.16 (0.449)
Lutjanus spp.							5.22	14.52	-9.30 **(<0.001)**	4.12	8.44	-4.31 **(<0.001)**
Lutjanus campechanus				0.36	0.66	-0.30 (0.270)	2.90	8.49	-5.59 **(<0.001)**	5.67	7.64	-1.97 (0.072)
Lutjanus griseus				0.36	0.17	0.18 (0.231)	1.74	2.01	-0.27 (0.359)	0.52	1.36	-0.84 (0.076)
Pristipomoides aquilonaris	2.60	2.00	0.60 (0.297)	0.36	1.05	-0.69 (0.129)	13.62	13.95	-0.33 (0.430)	5.67	6.51	-0.84 (0.251)
Rhomboplites aurorubens	1.30	1.72	-0.42 (0.344)	0.72	1.79	-1.07 (0.088)	32.46	25.96	6.51 **(0.003)**	14.95	13.36	1.58 (0.179)
Lobotes surinamensis				0.72	0.87	-0.16 (0.389)				5.41	4.59	0.83 (0.218)
Haemulidae				0.72	0.26	0.45 (0.069)	2.03	3.46	-1.43 (0.073)	0.26	1.08	-0.82 (0.059)
Cynoscion spp.	0.65	0.55	0.10 (0.434)	1.08	0.22	0.86 **(0.001)**	11.88	21.24	-9.36 **(<0.001)**	4.90	12.97	-8.07 **(<0.001)**
Sciaenops ocellatus							9.28	13.45	-4.18 **(0.012)**	4.12	10.65	-6.52 **(<0.001)**
Mullidae	19.48	11.15	8.33 **(0.001)**	75.63	50.87	24.75 **(<0.001)**	0.29	0.75	-0.46 (0.159)	6.70	6.00	0.70 (0.281)
Chaetodontidae	0.65	1.72	-1.07 (0.153)	0.36	2.58	-2.22 **(0.010)**	0.87	0.69	0.18 (0.345)	1.55	0.79	0.75 **(0.047)**
Pomacanthidae	0.65	2.27	-1.62 (0.088)	1.08	1.22	-0.15 (0.411)	0.29	0.82	-0.53 (0.138)	0.26	0.91	-0.65 (0.089)
Pomacentridae	1.95	8.05	-6.10 **(0.003)**	4.66	4.89	-0.23 (0.429)	8.70	8.36	0.34 (0.411)	4.38	8.49	-4.11 **(0.002)**
Labridae	24.68	31.11	-6.43 **(0.042)**	4.30	4.02	0.28 (0.405)	72.46	40.73	31.73 **(<0.001)**	14.95	8.55	6.40 **(<0.001)**
Scaridae	12.34	28.08	-15.74 **(<0.001)**	3.23	4.02	-0.79 (0.250)	20.58	22.31	-1.73 (0.220)	3.61	3.28	0.32 (0.360)

TABLE 3. Continued

Taxon	Spring Bongo			Spring Neuston			Fall Bongo			Fall Neuston		
	UNIS %FO n = 154	GOM %FO n = 1453	Diff. (p)	UNIS %FO n = 279	GOM %FO n = 2290	Diff. (p)	UNIS %FO n = 345	GOM %FO n = 1591	Diff. (p)	UNIS %FO n = 388	GOM %FO n = 1766	Diff. (p)
Acanthuridae	1.30	6.26	-4.96 (0.199)	0.36	1.66	-1.30 **(0.006)**				0.26	0.11	0.14 **(0.045)**
Trichiurus lepturus	8.44	3.44	5.00 **(<0.001)**	1.08	1.05	0.03 (0.482)	16.23	14.52	1.71 (0.183)	2.58	2.77	-0.20 (0.407)
Acanthocybium solandri							0.58	0.75	-0.17 (0.354)			
Katsuwonus pelamis	10.39	22.37	-11.98 **(<0.001)**	6.45	10.39	-3.94 **(0.016)**	5.80	5.15	0.64 (0.295)	2.32	2.60	-0.29 (0.362)
Scomberomorus cavalla							12.75	23.38	-10.63 **(<0.001)**	11.08	13.25	-2.17 (0.104)
Scomberomorus maculates	3.25	0.96	2.28 **(0.002)**	2.51	0.96	1.55 **(0.004)**	4.06	12.76	-8.70 **(<0.001)**	3.35	10.82	-7.46 **(<0.001)**
Thunnus spp.	12.99	22.02	-9.04 **(0.003)**	8.96	21.97	-13.00 **(<0.001)**	17.68	18.73	-1.05 (0.309)	16.75	21.74	-4.99 **(0.009)**
Thunnus thynnus	4.55	10.87	-6.33 **(0.006)**	6.81	10.44	-3.63 **(0.024)**						
Xiphius gladius				1.08	4.50	-3.42 **(0.003)**						
Istiophoridae	1.30	2.96	-1.66 (0.112)	3.94	9.43	-5.49 **(0.001)**	0.58	1.13	-0.55 (0.166)	5.93	9.74	-3.81 **(0.006)**
Peprilus alepidotus	0.65	0.28	0.37 (0.188)				10.14	13.14	-2.99 **(0.050)**	3.87	4.30	-0.44 (0.336)
Peprilus burti	1.30	1.79	-0.49 (0.323)	2.15	1.09	1.06 **(0.044)**	26.09	11.38	14.71 **(<0.001)**	4.38	1.59	2.80 **(<0.001)**

nets in the UNIS study area was 5% higher than Gulfwide, but in the fall was 2% lower than would be expected based on the number of survey samples taken in the study area. Observed differences in UNIS study area and Gulfwide survey total catch were even more pronounced for fish larvae collected in neuston samples. During spring surveys, the study area contributed only 12% of the total number of samples but larvae captured in neuston samples there accounted for 21% of the total standardized abundance Gulfwide. During fall surveys, the study area contributed 22% of the total number of samples but larvae captured in neuston samples there accounted for only 16% of the Gulfwide total.

Information is provided below on the number of larvae captured and frequency of occurrence (Table 1), mean abundance, mean CPUE, and distribution (Figures 5–62) and size for the early life stages of 61 taxa of fishes collected in bongo and neuston samples during SEAMAP spring and fall surveys within the UNIS study area. Also provided are comments on taxonomic resolution and relative ease or difficulty of larval stage identification for certain taxa. Percent frequency of occurrence (Table 3) and percent of total standardized abundance or total standardized CPUE, i.e. relative abundance of the select taxa, in the study area are compared to Gulfwide values (Table 4).

Taxon Specific Information

ELOPIDAE (7 occurrences; 16 larvae; Figure 5)

Elopomorph species have a leptocephalus larval form with a forked tail which distinguishes them from the leptocephali of spiny and true eels. It is likely that all these Elopomorph larvae were ladyfish, either *Elops saurus* or *E. smithi* (McBride et al. 2010). Other fork–tailed leptocephali from closely related families are tarpon (Megalopidae) and bonefish (Albulidae), which are morphologically distinct and rare in the northern GOM. Elopid leptocephali, 13.5–29.2 mm BL (n = 8 specimens measured), occurred only 7 times, all during fall surveys, with captures almost equally divided between bongo (3) and neuston (4) net samples. Eleven of the 16 specimens captured were taken in neuston collections (Table 1). The stations where elopid larvae were found in bongo samples and in 3 of the 4 neuston samples were located along or west of 87°W longitude (Figure 5). Percent occurrence of elopid larvae in the UNIS study area did not differ significantly from their occurrence Gulfwide (Table 3). Relative abundance and CPUE in the 2 areas were similar,
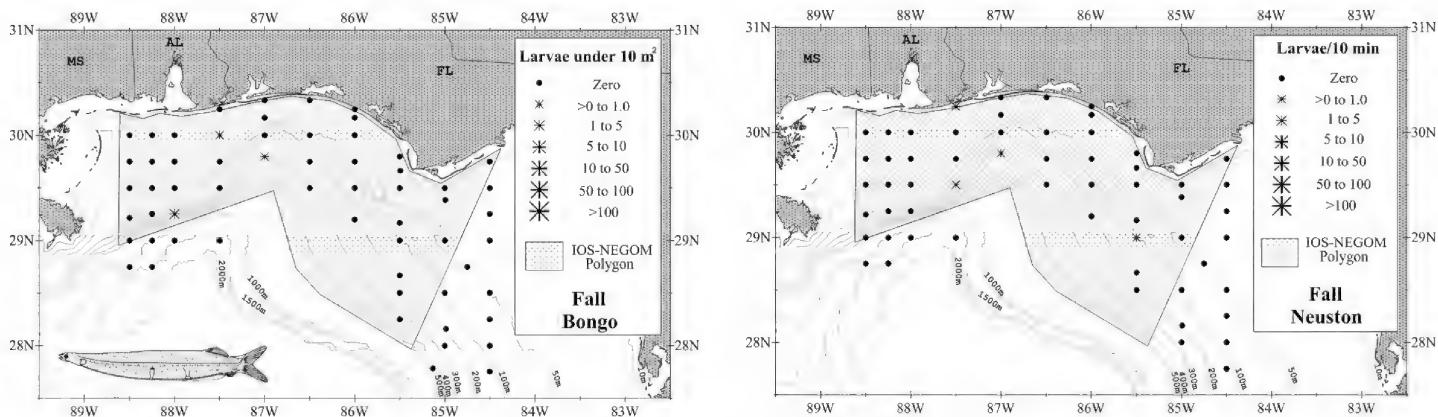
Figure 5. *Mean abundance and mean CPUE of ladyfish (Elopidae) larvae at stations in the UNIS study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*
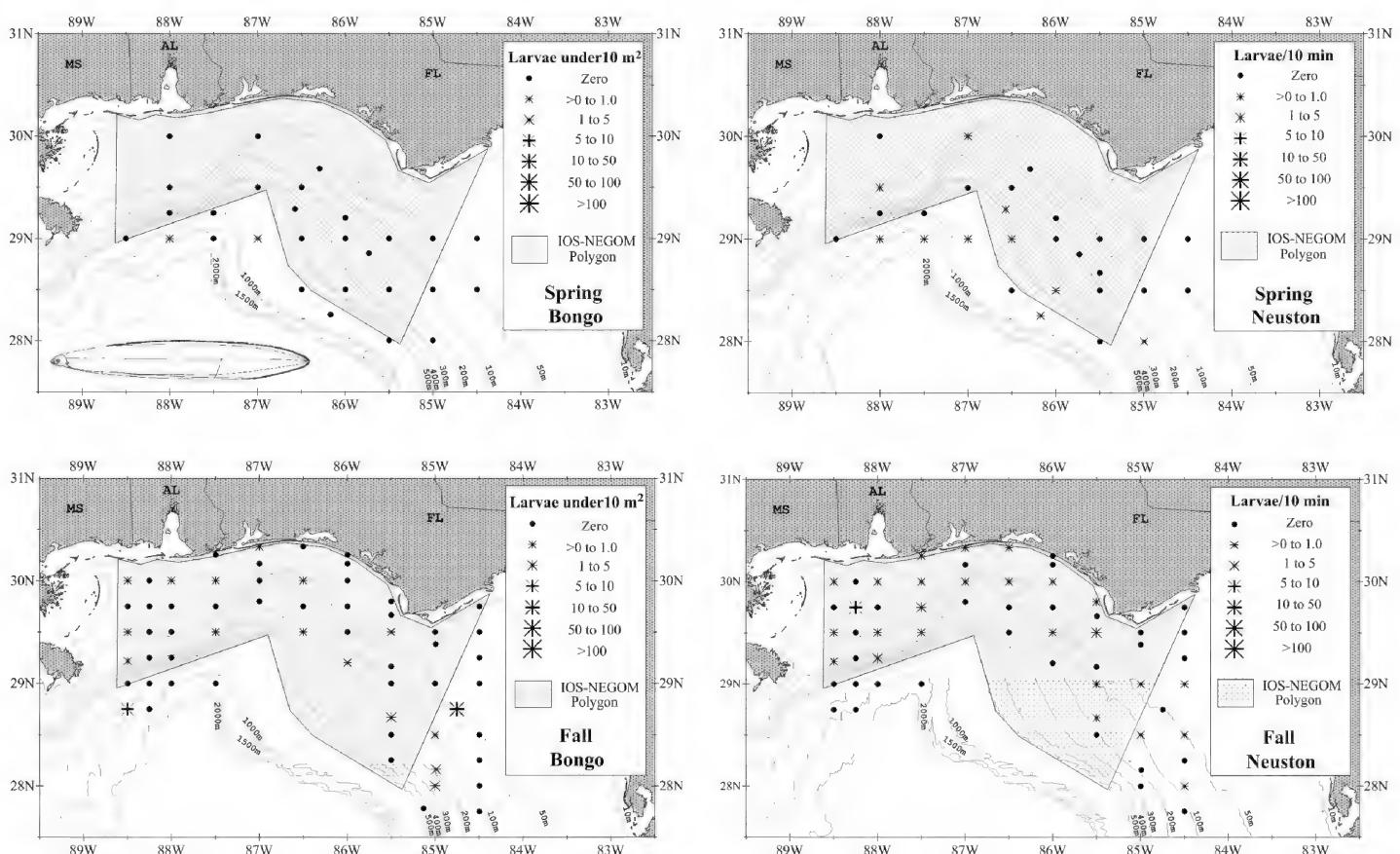

Figure 6. *Mean abundance and mean CPUE of moray eel (Muraenidae) larvae at stations in the UNIS study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

differing by < 0.1% (Table 4).

MURAENIDAE (83 occurrences; 188 larvae; Figure 6)

Moray eels are a characteristic component of hard or live bottom communities throughout the GOM. Their leptocephalus larvae can be distinguished from the young of other eel families by the complete absence or greatly reduced state of their pectoral fins. Moray eel leptocephali, 3.5–107 mm BL (n = 65), occurred more frequently in fall surveys than in spring (71 vs. 12 occurrences) and were captured more

often in neuston than in bongo samples (60 vs. 23, Table 1). These larvae were found, for the most part, at the most offshore stations during spring surveys but were dispersed throughout the study area during fall surveys, from the closest inshore to the farthest offshore stations (Figure 6). During spring surveys muraenid eel larvae were as common in UNIS collections as they were throughout the GOM; i.e. there was no significant difference in occurrence between the 2 areas. In fall surveys, however, muraenid eel larvae oc-

TABLE 4. Relative abundance of 61 select ichthyoplankton taxa captured in USGS Northeastern Gulf of Mexico Ichthyoplankton Synopsis (UNIS) study area samples alone and in all Gulfwide (GOM) survey samples combined during Southeast Area Monitoring and Assessment Program (SEAMAP) spring and fall plankton surveys over the period 1982-1999. PTA = percent of total abundance in bongo samples. PTC = percent of total CPUE in neuston samples. n = number of samples.

| | Spring survey | | | | Fall survey | | | |
| | Bongo PTA | | Neuston PTC | | Bongo PTA | | Neuston PTC | |
Taxon	UNIS n = 154	GOM n = 1453	UNIS n = 279	GOM n = 2290	UNIS n = 345	GOM n = 1591	UNIS n = 388	GOM n = 1766
Elopidae	0	<0.01	0	0.01	0.01	<0.01	0.02	0.01
Muraenidae	0.01	0.02	0.05	0.07	0.03	0.03	0.25	0.14
Engraulidae	1.66	1.34	5.98	3.27	14.12	10.13	40.75	28.14
Etrumeus teres	0.88	0.39	2.31	0.57	<0.01	<0.01	<0.01	<0.01
Harengula jaguana	0.15	0.04	5.78	1.71	0.08	0.31	2.43	3.31
Opisthonema oglinum	0.09	0.03	0.47	0.13	0.42	4.04	0.31	7.52
Sardinella aurita	<0.01	0.26	0.08	0.99	1.16	3.42	4.77	6.67
Sternoptychidae	6.02	4.47	0.29	0.09	1.64	0.96	<0.01	<0.01
Synodontidae	2.34	2.84	0.88	0.86	6.48	4.86	0.66	0.69
Paralepididae	2.11	1.75	0.05	0.11	0.37	0.23	0.01	<0.01
Carapidae	0.02	0.01	0	0	0.08	0.03	<0.01	<0.01
Carapus bermudensis	0.03	0.03	0	<0.01	0.25	0.12	0	<0.01
Bregmacerotidae	7.03	8.09	0.75	0.69	8.47	8.30	0.24	0.08
Mugilidae	0.09	0.08	3.91	5.33	<0.01	<0.01	0.03	0.40
Melamphaidae	0.18	0.36	0.01	0.02	0.03	0.03	<0.01	<0.01
Holocentridae	<0.01	0.07	0.04	0.52	0.01	0.02	0.02	0.05
Serranidae	0.63	1.29	0.14	0.39	1.01	0.84	0.14	0.09
Serraninae	0.13	0.27	0.45	0.39	0.87	0.67	0.65	0.72
Anthiinae	0.18	0.45	0.13	0.18	0.09	0.04	0.01	0.01
Epinephelinae	0.01	0.03	0	<0.01	0	<0.01	0	0
Grammistinae	0.04	0.04	0.07	0.05	0.13	0.10	0.08	0.09
Priacanthidae	0.08	0.06	0.07	0.12	0.09	0.11	0.19	0.26
Apogonidae	0.09	0.13	0.35	0.24	0.29	0.40	0.17	0.29
Rachycentron canadum	0	<0.01	0.04	0.02	0	<0.01	0.01	0.04
Coryphaenidae	0.06	0.10	0.63	1.29	0.02	0.01	0.25	0.14
Caranx spp.	0.17	0.54	1.86	5.77	0.04	0.07	1.06	0.75
Chloroscombrus chrysurus	0	<0.01	0.02	0.03	0.82	4.45	2.96	9.36
Decapterus spp.	0.05	0.14	0.92	0.65	3.46	1.87	5.15	2.56
Selar crumenophthalmus	0.01	0.03	<0.01	0.12	0.13	0.22	1.03	0.37
Selene spp.	0	<0.01	0	0.01	0.04	0.11	0.04	0.07
Seriola spp.	0.03	0.04	0.91	0.77	<0.01	0.01	0.13	0.10
Trachinotus spp.	0	<0.01	0.15	0.17	0	<0.01	0.04	0.04
Trachurus lathami	0.05	0.03	0.11	0.11	0	<0.01	0	<0.01
Lutjanidae	0.02	0.06	0.01	0.04	0.66	0.71	0.10	0.10
Lutjanus spp.	0	0.01	0	0.02	0.03	0.11	0.07	0.27
Lutjanus campechanus	0	<0.01	<0.01	0.03	0.01	0.05	0.10	0.15
Lutjanus griseus	0	<0.01	<0.01	<0.01	0.01	0.01	<0.01	0.02
Pristipomoides aquilonaris	0.01	0.02	<0.01	0.03	0.15	0.17	0.14	0.14
Rhomboplites aurorubens	0.01	0.01	0.05	0.10	0.31	0.26	0.54	0.50
Lobotes surinamensis	0	<0.01	<0.01	0.01	0	<0.01	0.06	0.03
Haemulidae	0	<0.01	<0.01	0.01	0.13	0.07	<0.01	0.02
Cynoscion spp.	<0.01	0.01	0.04	0.01	0.27	0.96	0.15	2.16
Sciaenops ocellata	0	0	0	<0.01	0.14	0.40	0.19	1.04
Mullidae	0.26	0.27	47.36	26.96	<0.01	<0.01	0.36	0.16
Chaetodontidae	<0.01	0.01	<0.01	0.06	0.01	<0.01	0.01	0.01
Pomacanthidae	<0.01	0.02	0.01	0.02	<0.01	<0.01	<0.01	0.01
Pomacentridae	0.12	0.12	0.09	0.18	0.05	0.05	0.07	0.11
Labridae	0.30	0.64	0.14	0.14	3.67	1.22	0.32	0.15
Scaridae	0.08	0.70	0.03	0.12	0.36	0.38	0.05	0.03
Acanthuridae	0.01	0.08	<0.01	0.03	0	0.01	<0.01	<0.01
Trichiurus lepturus	0.13	0.07	0.03	0.03	0.20	0.16	0.04	0.03
Acanthocybium solandri	0	<0.01	0	<0.01	<0.01	<0.01	0	<0.01
Katsuwonus pelamis	0.07	0.28	0.10	0.36	0.05	0.05	0.06	0.05
Scomberomorus cavalla	0	<0.01	0	0.03	0.06	0.19	0.16	0.30
Scomberomorus maculatus	0.03	0.01	0.18	0.06	0.02	0.12	0.05	0.50
Thunnus spp.	0.20	0.40	0.36	1.29	0.18	0.27	0.67	0.99
Thunnus thynnus	0.04	0.21	0.30	0.71	0	0	0	0
Xiphias gladius	0	<0.01	0.01	0.07	0	0	0	<0.01
Istiophoridae	0.02	0.03	0.05	0.30	<0.01	<0.01	0.09	0.16
Peprilus alepidotus	<0.01	<0.01	0	<0.01	0.10	0.10	0.08	0.08
Peprilus burti	0.01	0.02	0.04	0.02	0.76	0.22	0.14	0.03
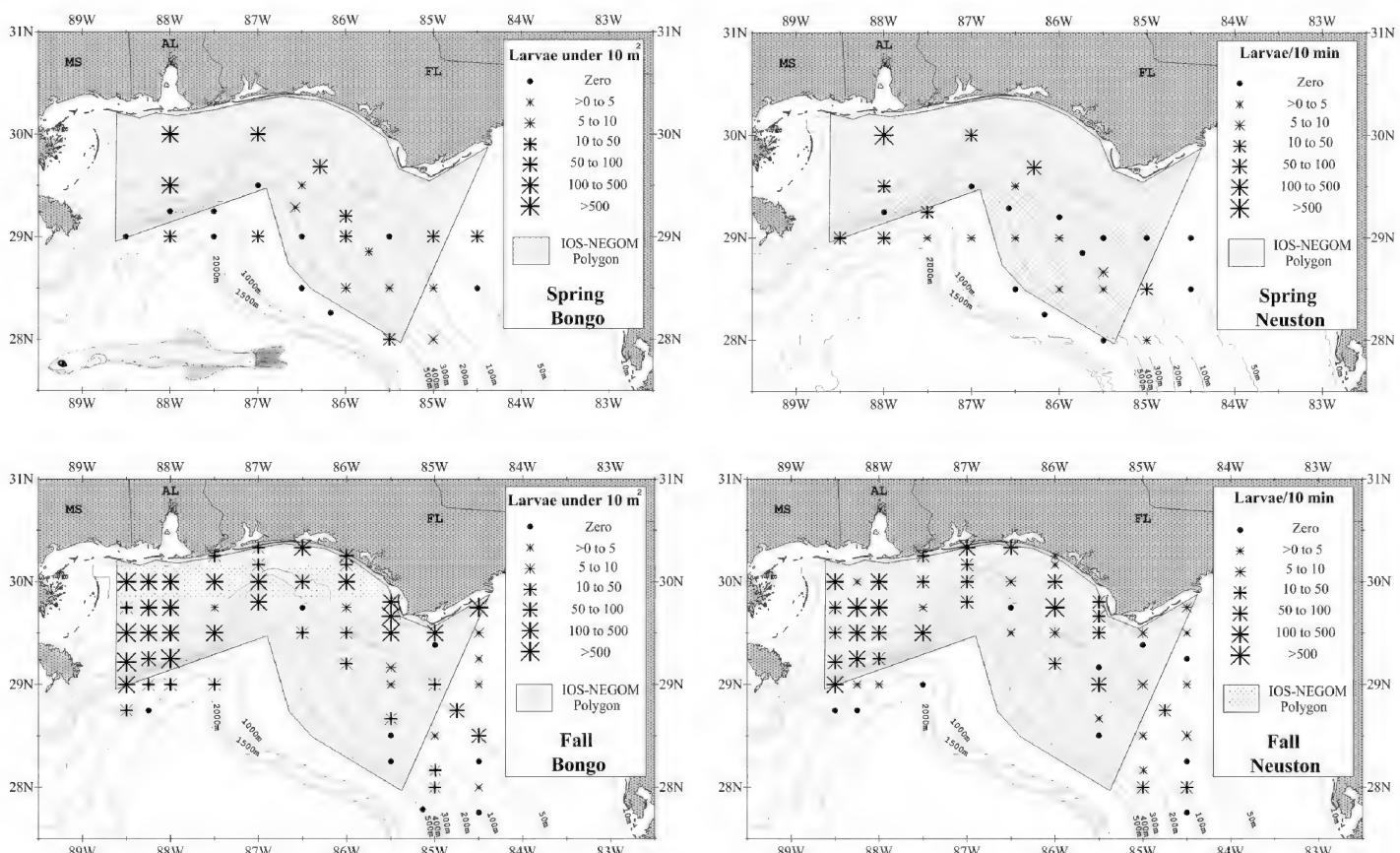
Figure 7. *Mean abundance and mean CPUE of anchovy (Engraulidae) larvae at stations in the UNIS study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

curred significantly more often in UNIS neuston collections than in neuston collections Gulfwide (Table 3). The relative abundance and CPUE in the 2 areas were similar, differing by < 0.2% (Table 4).

ENGRAULIDAE (573 occurrences; 40,732 larvae; Figure 7)

Anchovy larvae were the most frequently caught and most numerous of all fish larvae in UNIS study area collections. At least 6 species of engraulids are known to occur over the continental shelf in the NEGOM but their larvae cannot be easily distinguished from one another until later in the juvenile stage (Farooqi et al. 2006). All but 97 of 573 occurrences took place during fall surveys with specimens taken in fall collections outnumbering those in spring collections by an order of magnitude (Table 1). The overall size range of larvae captured in UNIS study area collections was 1.2–34 mm BL (n= 678). Anchovy larvae were taken as frequently in bongo as in neuston samples but larvae were somewhat more numerous in neuston samples. As would be expected, anchovy larvae occurred throughout the survey area but highest mean abundances were consistently observed in the westernmost region from nearshore to the most offshore stations and elsewhere in the study area inshore of the 50 m contour (Figure 7). During both seasons, anchovy larvae occurred significantly more often in UNIS study area samples than in Gulfwide samples (Table 3). Relative abundance and CPUE

in spring samples (both bongo and neuston) and fall bongo samples in the 2 areas differed by < 5% (Table 4), whereas, the relative CPUE in UNIS study area fall neuston samples exceeded the Gulfwide value by 12%. This was disproportionately higher than expected given the fewer samples taken in the study area.

CLUPEIDAE

Etrumeus teres (56 occurrences; 1,306 larvae; Figure 8)

Larvae of the round herring were found in the study area almost exclusively during spring surveys. Incidence of capture in bongo and neuston samples was similar but larvae were nearly 3 times more numerous in neuston than in bongo collections (Table 1). The overall size range of larvae captured in UNIS study area collections was 3.0–18.0 mm BL (n= 144). Most occurrences and the highest mean abundances were observed at stations along or east of 87°W longitude (Figure 8). Round herring were taken significantly more often in UNIS samples than in Gulfwide samples during spring surveys but at comparable frequencies (i.e., no statistical difference) during fall surveys (Table 3). The relative abundances and CPUEs in the two areas differed by < 2% during spring surveys and < 0.01% during fall surveys (Table 4).

Harengula jaguana (137 occurrences; 3,909 larvae; Figure 9)

Scaled sardine larvae ranked second in occurrence and abundance among the clupeid larvae identified to species.
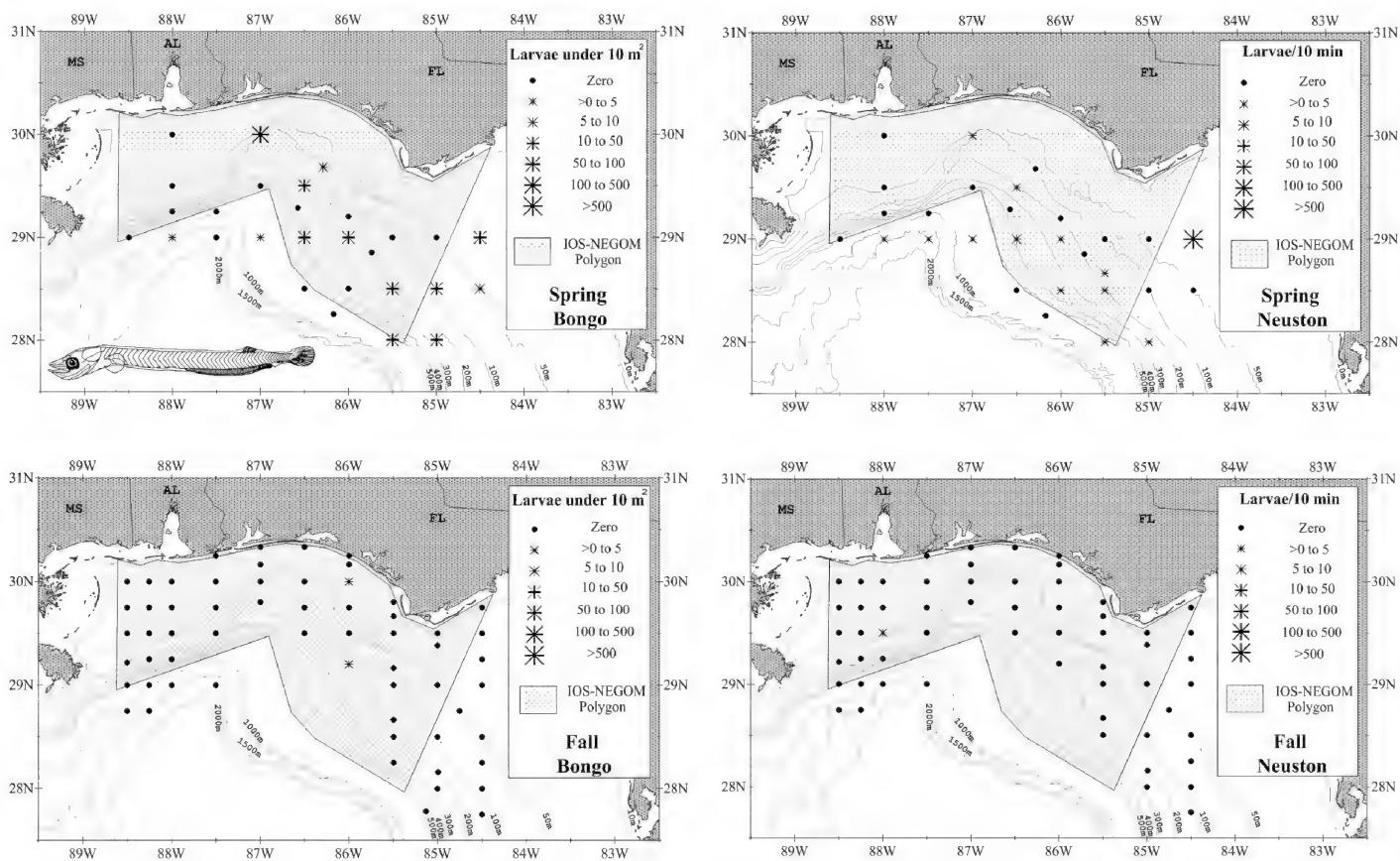
Figure 8. *Mean abundance and mean CPUE of round herring, Etrumeus teres, larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*
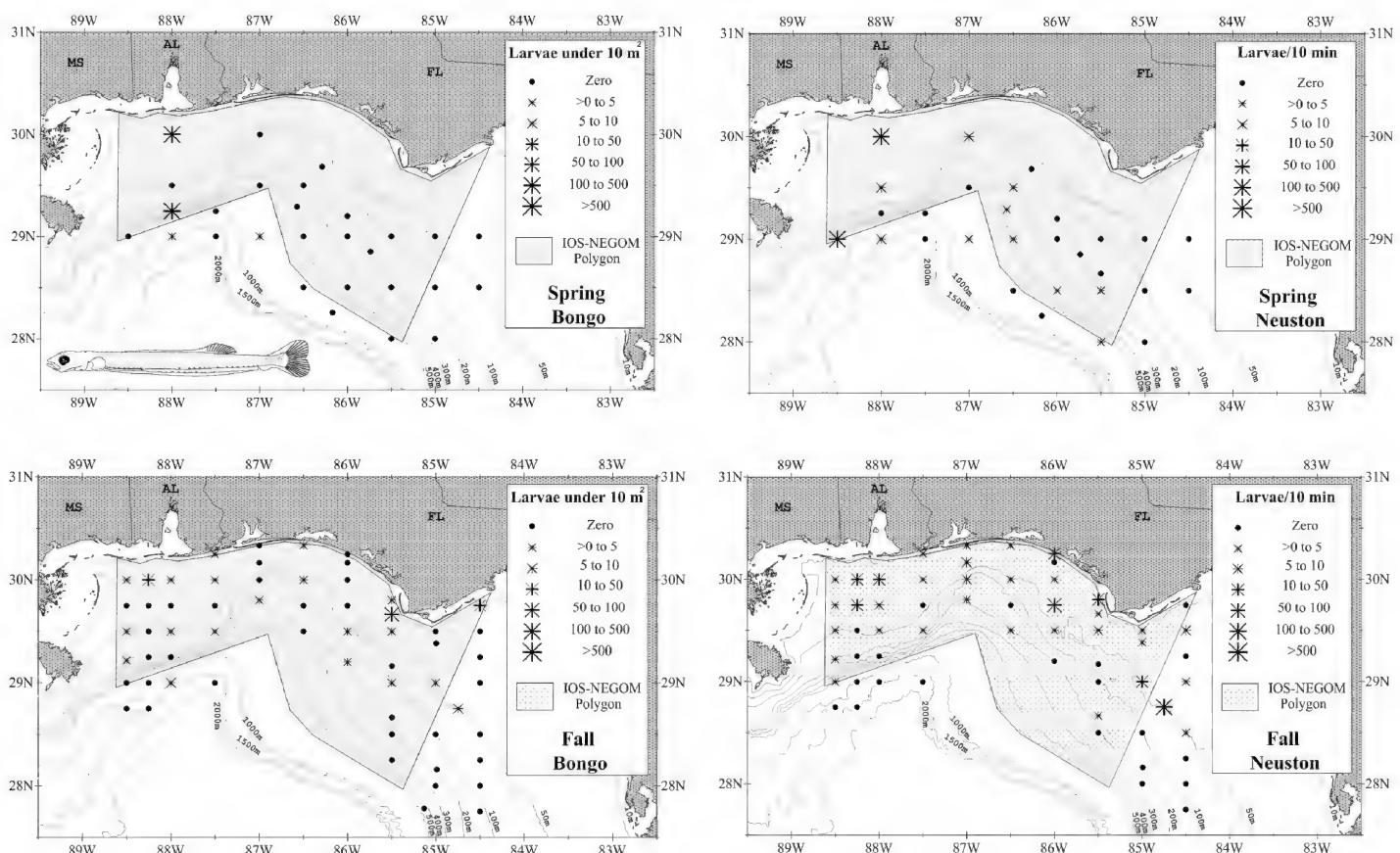

Figure 9. *Mean abundance and mean CPUE of scaled sardine, Harengula jaguana, larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*
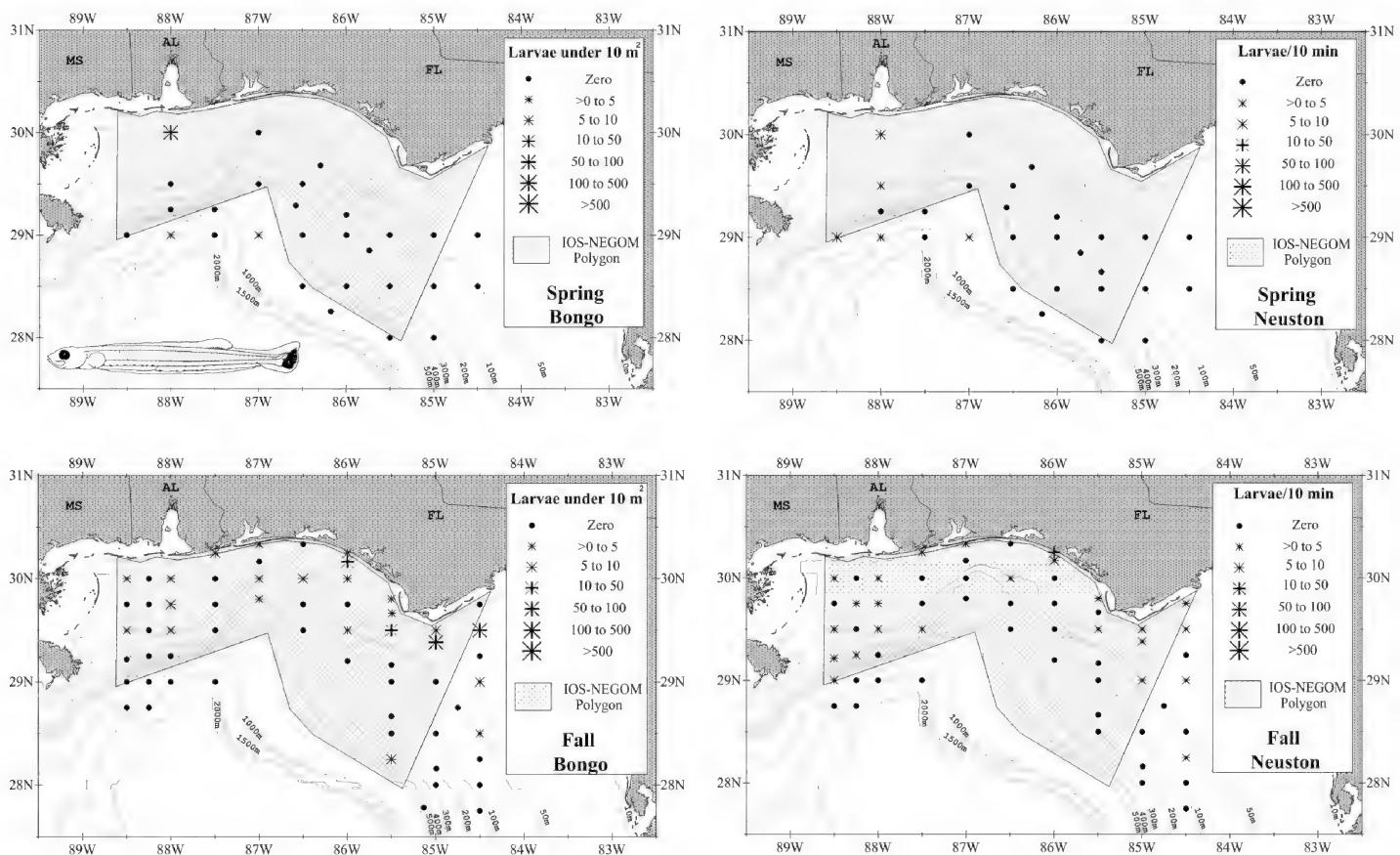
Figure 10. *Mean abundance and mean CPUE of thread herring,* Opisthonema oglinum, *larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

Although fall survey collections accounted for two–thirds of their occurrences, nearly twice as many scaled sardine larvae were taken during spring surveys (Table 1). The overall size range of larvae captured in UNIS study area collections was 3.0– 21.0 mm BL (n= 250). Larvae were captured 3.5 times more often and were an order of magnitude more numerous in neuston than bongo net samples. In the spring, highest mean abundances of scaled sardine larvae were observed west of 87°W longitude while during fall surveys larvae were distributed across the study area with greatest mean abundances inshore of the 50 m contour (Figure 9). During spring surveys scaled sardine larvae were taken significantly more often in the study area than Gulfwide, but the reverse was true during fall surveys when larvae were significantly less common in study area samples than Gulfwide (Table 3). The relative abundances and CPUEs in the two areas differed by < 5% (Table 4).

Opisthonema oglinum (89 occurrences; 1,163 larvae; Figure 10)

Atlantic thread herring larvae were more numerous and occurred more frequently in fall than in spring surveys. Unlike the other abundant clupeid larvae, they were more frequently taken and more numerous in bongo than in neuston net samples (Table 1). The overall size range of larvae cap-

tured in UNIS study area collections was 1.2–25.5 mm BL (n= 299). Larvae were not found east of 87° W longitude during spring surveys but occurred throughout the study area during fall surveys when most occurrences and highest mean abundances were observed inshore of the 50 m contour and east of 87°W longitude (Figure 10). During spring surveys, Atlantic thread herring larvae were taken significantly more often in the study area than Gulfwide but the reverse was true during fall surveys when larvae were significantly less common in study area samples than Gulfwide (Table 3). The relative abundance during spring surveys and fall bongo samples in the two areas was similar, differing by < 5% (Table 4). However, relative CPUE of threadfin herring larvae was 7% lower than Gulfwide in fall neuston samples.

Sardinella aurita (148 occurrences; 4,360 larvae; Figure 11)

Larvae of the Spanish sardine were the most frequently taken and most abundant larval clupeid in the study area. Larvae were taken almost exclusively during fall surveys and were comparably represented in bongo and neuston samples (Table 1). Mean size in bongo samples was 6.5 mm BL (n = 171; range = 2.7–16.0 mm) and mean size in neuston samples was 10.1 mm BL (n = 256; range = 1.5–74 mm). Larvae occurred most frequently and in highest abundance at stations east of 87° W longitude and generally over depths ≤100
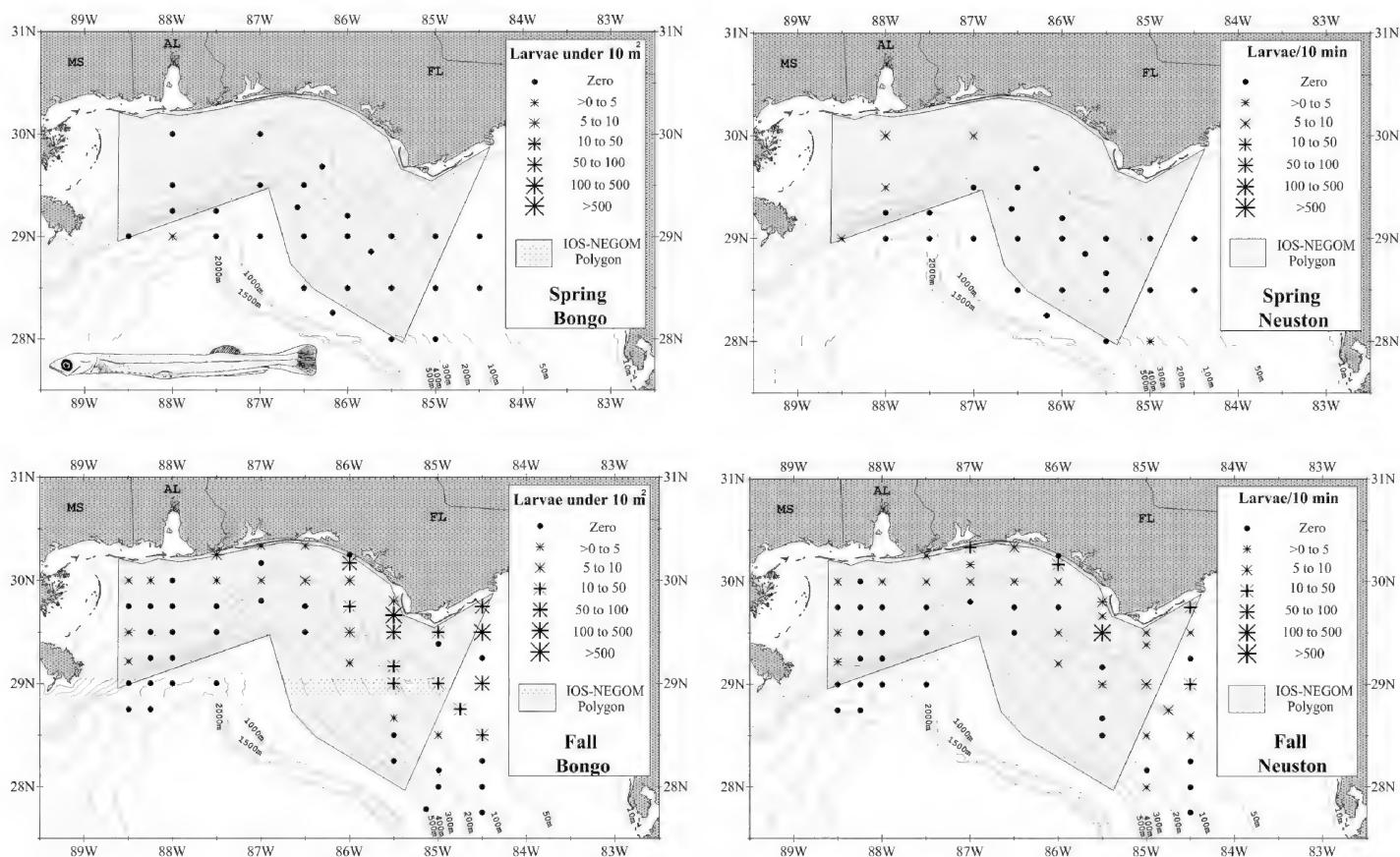
Figure 11. *Mean abundance and mean CPUE of Spanish sardine,* Sardinella aurita, *larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

m (Figure 11). Spanish sardine larvae occurred as frequently in the UNIS study area as Gulfwide (Table 3). Relative abundance and CPUE in the 2 areas differed by < 2% (Table 4).

STERNOPTYCHIDAE (210 occurrences; 3,533 larvae; Figure 12)

Larvae of the bioluminescent meso– and bathypelagic hatchet fishes were fairly numerous in UNIS study area collections during both spring and fall surveys (Table 1). The overall size range of larvae captured in UNIS study area collections was 1.7–29.5 mm BL (n = 115). Hatchet fish larvae were taken almost exclusively in bongo samples, at stations beyond 50 m, and along the contours outlining the DeSoto Canyon (Figure 12). Mean abundances at offshore stations consistently ranged from 10–100 larvae/10 m^2. Hatchet fish larvae occurred significantly more often in bongo samples in the study area than Gulfwide during both spring and fall surveys but were found at comparable frequencies in both areas in spring and fall neuston samples (Table 3). Relative abundances and CPUEs in the 2 areas were similar differing by < 2% (Table 4).

SYNODONTIDAE (501 occurrences; 7,229 larvae; Figure 13)

The lizardfishes are an important group of benthic predatory fishes common on soft bottom substrates of the continental shelf in the GOM and are considered an important member of halo communities extending away from reefs. The larvae of this family of 6 GOM species were among the most frequently taken and numerous larvae in SEAMAP plankton collections. Most lizardfish larvae were taken in bongo net samples during fall surveys and were collected at all but 3 UNIS stations (Table 1, Figure 13). The overall size range of larvae captured in UNIS study area collections was 1.3–43 mm BL (n = 545). The highest mean abundances were found at stations between the 50 and 200 m contours. Lizardfish larvae were captured significantly more often in the study area bongo samples than Gulfwide in both spring and fall. However, neuston–caught lizardfish larvae were found significantly more often in the UNIS study area only in the spring (Table 3). Relative abundances and CPUEs in the 2 areas were similar, differing by < 2% (Table 4).

PARALEPIDIDAE (215 occurrences; 1,053 larvae; Figure 14)

The barracudinas occur in epi–, meso–, and bathypelagic zones of the GOM and their larvae were not uncommon in the UNIS study area, occurring most often and in greatest numbers in bongo net collections during spring surveys (Table 1). The overall size range of larvae captured in UNIS study area collections was 2.0–31 mm BL (n = 181). The distribution of barracudina larvae, like hatchet fish larvae, followed the isobaths outlining the DeSoto Canyon across the full east–west extent of the study area (Figure 14). Mean abundances in bongo samples at most stations beyond the 100 m contour were typically between 10–50 larvae under

60
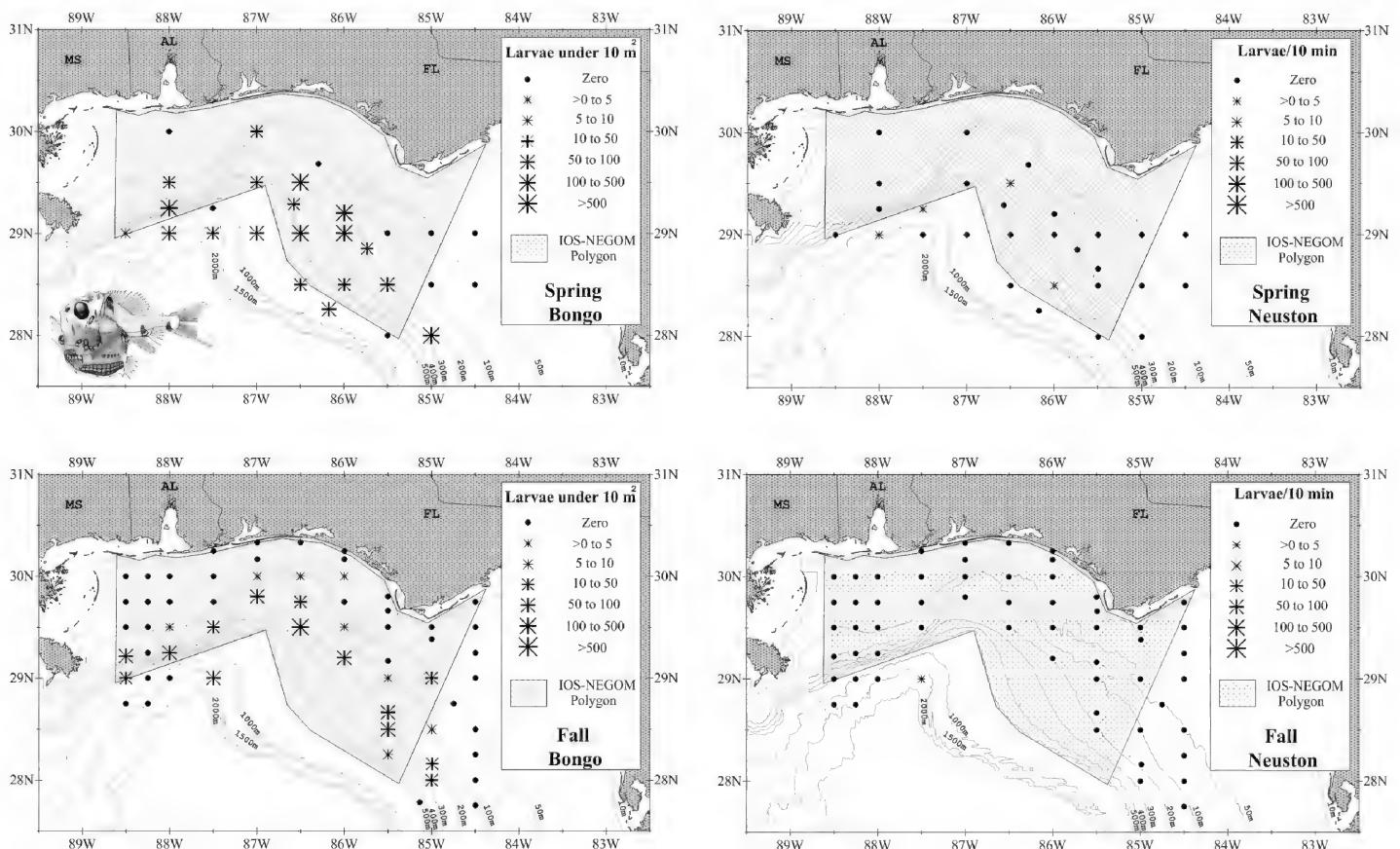
Figure 12. *Mean abundance and mean CPUE of hatchetfish (Sternoptychidae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*
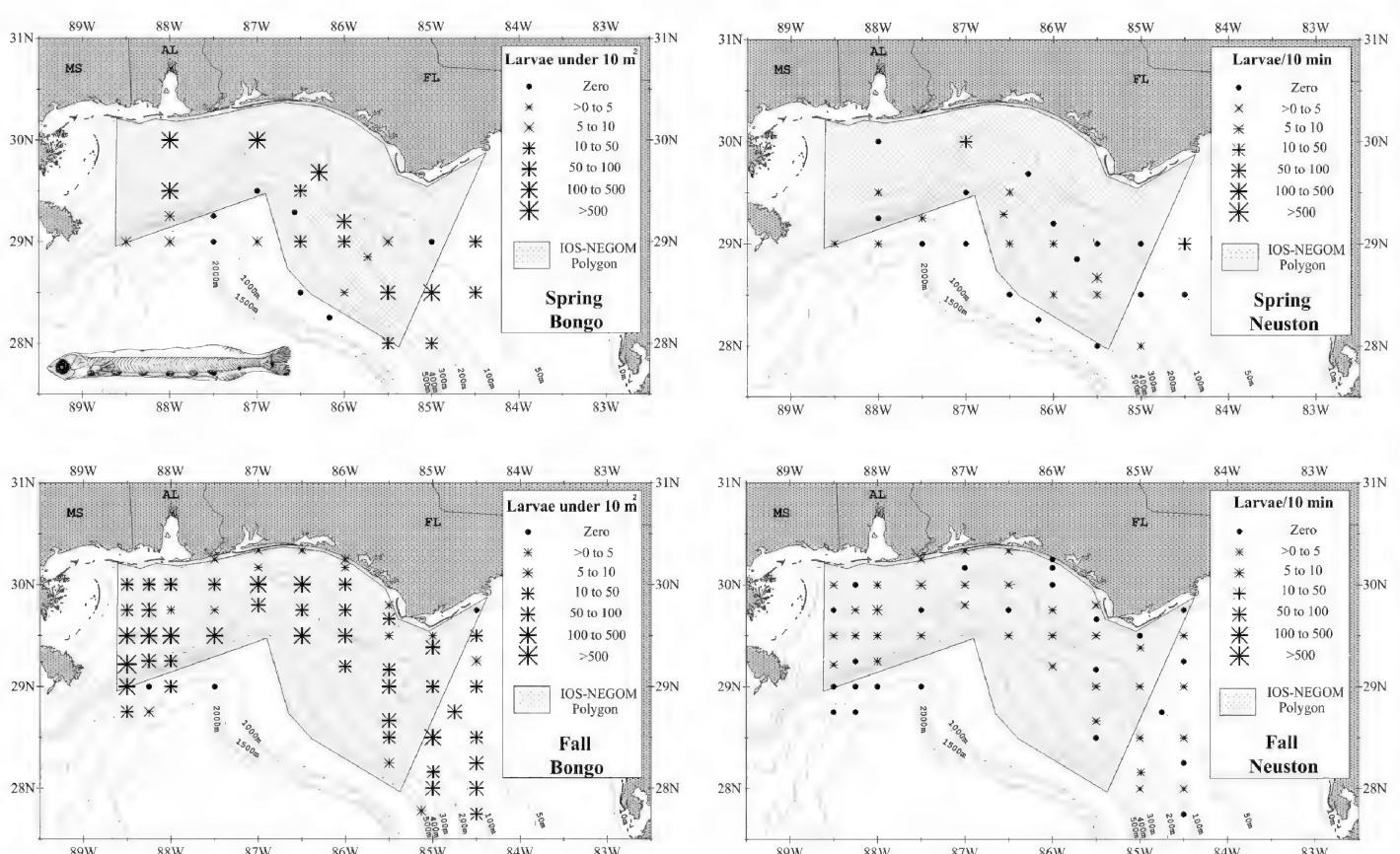
Figure 13. *Mean abundance and mean CPUE of lizardfish (Synodontidae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*
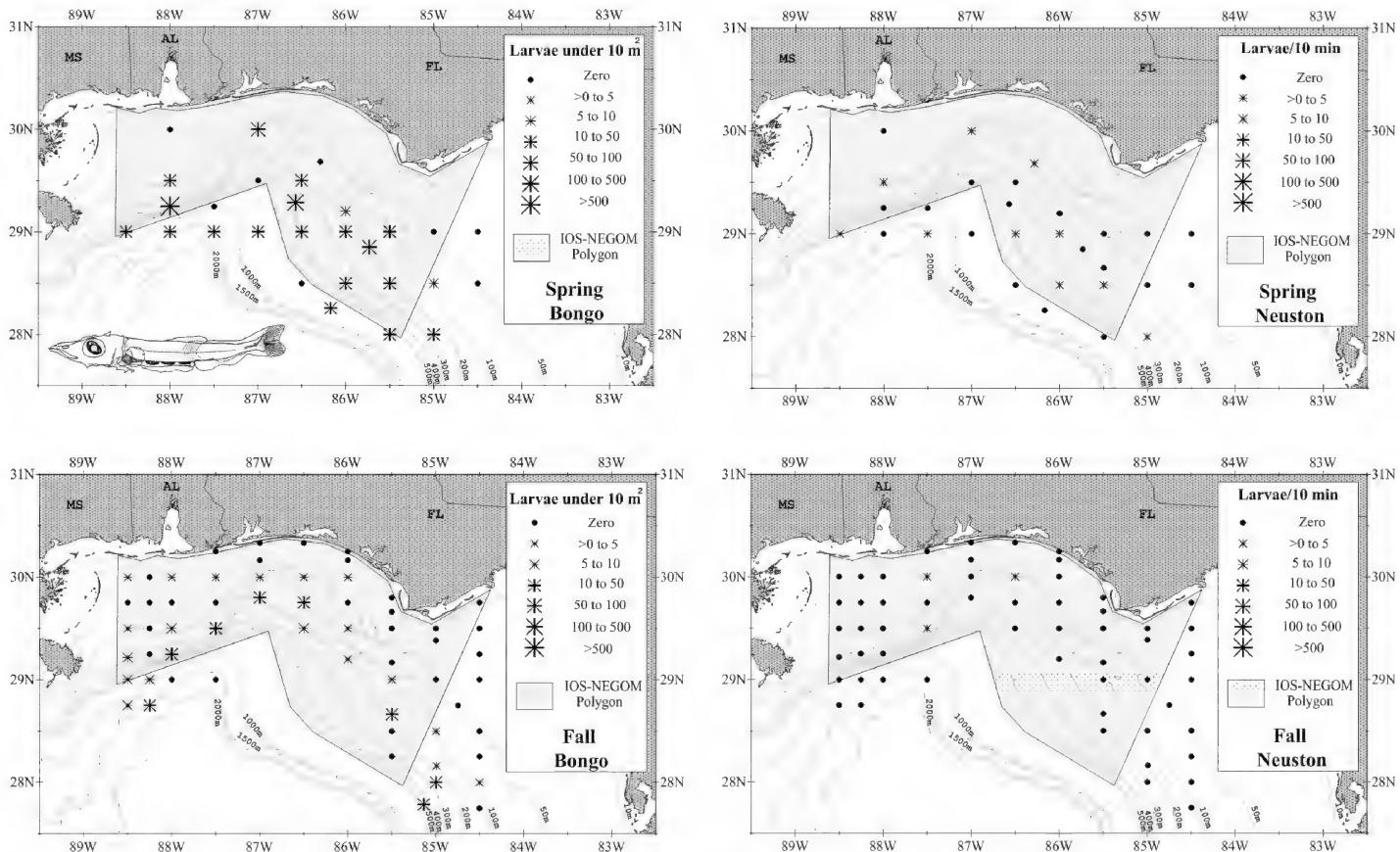
Figure 14. Mean abundance and mean CPUE of barracudina (Paralepididae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.

10 m^2. Barracudina larvae were captured significantly more often in the study area than Gulfwide only in the fall, but occurrence in spring samples was comparable in the 2 areas (Table 3). Relative abundances and CPUEs in the 2 areas were similar, differing by < 0.5% (Table 4).

CARAPIDAE (22 occurrences; 62 larvae; Figure 15)

Planktonic larvae of the pearlfishes are distinctive in having a vexillum or long, thread—like process anterior to the dorsal fin. Most species live as inquilines in the body cavity of certain benthic invertebrates but have free living planktonic larvae. Three species occur in the western central Atlantic; *Carapus bermudensis*, *Echiodon dawsonii*, and *Snyderidia canina*. The last species is thought to be free—living, but has been observed sheltering beneath the recumbent spines of the large, white, deepwater 'pancake' urchin, *Araeosoma* sp. (K. Sulak, unpublished data, USGS, Gainesville, FL). Pearlfish larvae not identified as *C. bermudensis* (see below) were captured almost exclusively in bongo collections during fall surveys (Table 1). The overall size range of pearlfish larvae captured in UNIS study area collections was 2.6—56 mm BL (n = 27). Highest mean abundances were found at stations in the southwest and southeast corners of the IOS—NEGOM research polygon in bongo samples during the fall (Figure 15). Three occurrences in spring bongo samples (not shown in Figure 15) were located in the same locality as the captures

in fall samples. Pearlfish larvae were captured significantly more often in the study area than Gulfwide during fall surveys but at comparable frequencies in spring bongo samples (Table 3). Relative abundances and CPUEs in the 2 areas were similar, differing by < 0.1% (Table 4).

Carapus bermudensis (68 occurrences; 210 larvae; Figure 16)

Larvae of this species of pearlfish were taken more frequently and in greater numbers during fall than spring surveys in bongo samples; larvae were never taken in neuston collections (Table 1). Mean size of captured *C. bermudensis* larvae was 14.2 mm BL (n = 56; range = 2.9—74 mm). Most captures of pearlfish larvae were made at stations east of 87°W longitude and generally over water depths ≤ 50 m (Figure 16). This was unlike the pattern among larvae identified only to the family level which were captured somewhat more often in the southwestern corner of the IOS—NEGOM research polygon (Figure 15). *Carapus bermudensis* larvae were captured significantly more often in the study area than in Gulfwide samples during fall surveys (Table 3). Occurrence during spring surveys and relative abundance (differing by < 0.1%) were comparable in the 2 areas (Table 4).

BREGMACEROTIDAE (441 occurrences; 9,918 larvae; Figure 17)

Codlets are generally known as epipelagic planktivores, but have been documented feeding epibenthically and intensely on reef—top habitat at night on NEGOM "Pinna-

62
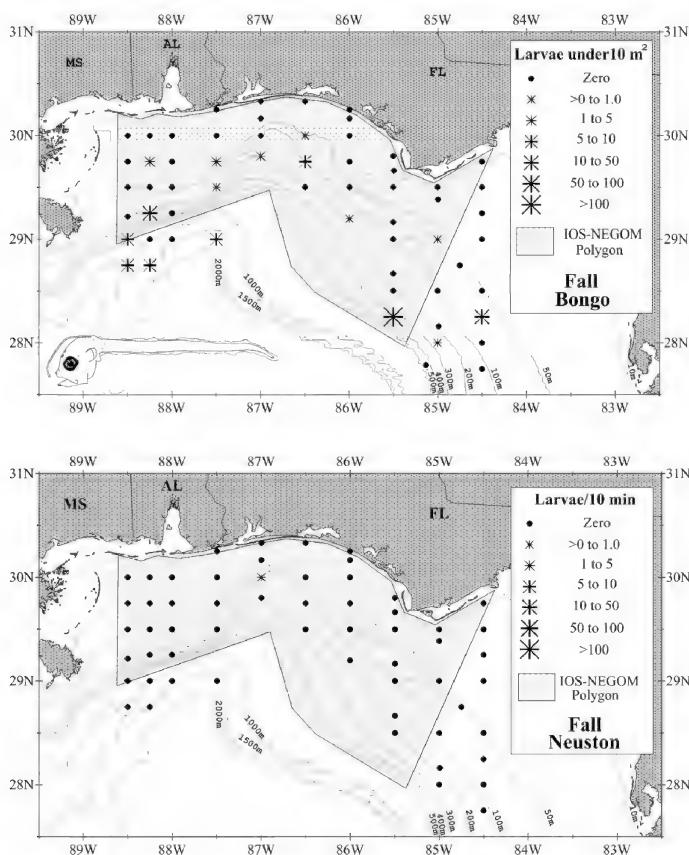
Figure 15. *Mean abundance and mean CPUE of the pearlfish (Carapidae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. (Map of 3 occurrences in spring survey bongo samples is not shown.) UNIS and SEAMAP defined in Figure 1.*
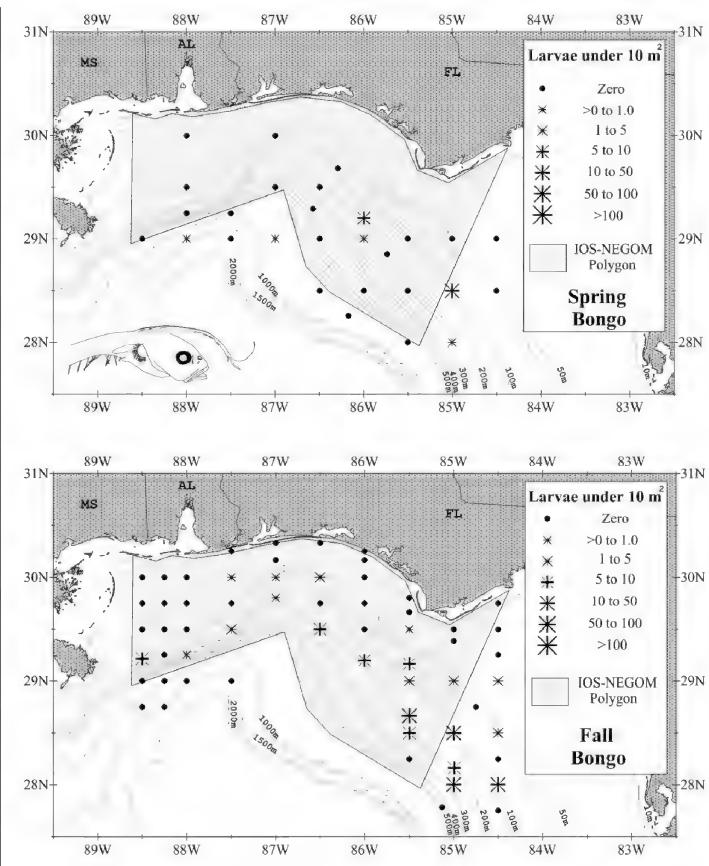
Figure 16. *Mean abundance of pearlfish, Carapus bermudensis, larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

cles Tract" reefs (K. Sulak, unpublished video data, USGS, Gainesville, FL). Codlet larvae were common throughout the UNIS study area during both spring and fall but most larvae were taken in bongo samples during fall surveys (Table 1). The overall size range of codlet larvae captured in UNIS study area collections was 1.6–28.3 mm BL (n = 493). Mean abundances of > 100 larvae under 10 m² were typical especially in the southwestern corner of the study area, the head of the DeSoto Canyon, and its eastern 'slopes' (Figure 17). Codlet larvae were taken in over 80% of bongo samples in the spring and over 55% in the fall in both the UNIS study area and Gulfwide. The occurrence of codlet larvae in neuston samples was much lower than in bongo samples with values never exceeding 15% during either spring or fall surveys. The frequency of capture in UNIS study area samples was significantly greater than in Gulfwide survey samples during both seasons except in fall neuston samples when frequency of capture was comparable in the 2 areas (Table 3). Relative abundances and CPUEs in both areas were similar, differing by < 2% (Table 4).

MUGILIDAE (154 occurrences; 1,669 larvae; Figure 18)

As silvery, pelagic juveniles, mullet inhabit surface waters of the open ocean for up to several months before migrating

inshore. Young of the abundant species of mullets in the GOM are likely to be present in UNIS study area plankton collections despite their differing spawning seasons; *Mugil cephalus* spawns in late fall and winter while M. *curema* spawns in the spring (Ditty and Shaw 1996). Young mullet were taken almost exclusively in neuston samples during spring surveys throughout the study area (Table 1, Figure 18). The overall size range of young mullet captured in UNIS study area collections was 2.1–11.0 mm BL (n = 16) in bongo samples and 3.0–27.5 mm BL (n = 302) in neuston samples. Springtime occurrence of young mullet in the UNIS study area neuston samples exceeded their occurrence in Gulfwide samples, 47% vs. 31%, but frequency of capture in spring bongo samples was comparable in the 2 areas (Table 3). The difference in percent occurrence between the 2 areas was statistically significant (Table 3). Larvae were significantly less common in the study area than Gulfwide during fall surveys. Relative abundances and CPUEs in both areas were similar, differing by < 2% (Table 4).

MELAMPHAIDAE (58 occurrences; 90 larvae; Figure 19)

The melamphaids are meso– and bathypelagic fishes whose larvae were taken in the UNIS study area almost exclusively in bongo samples, and were more common during
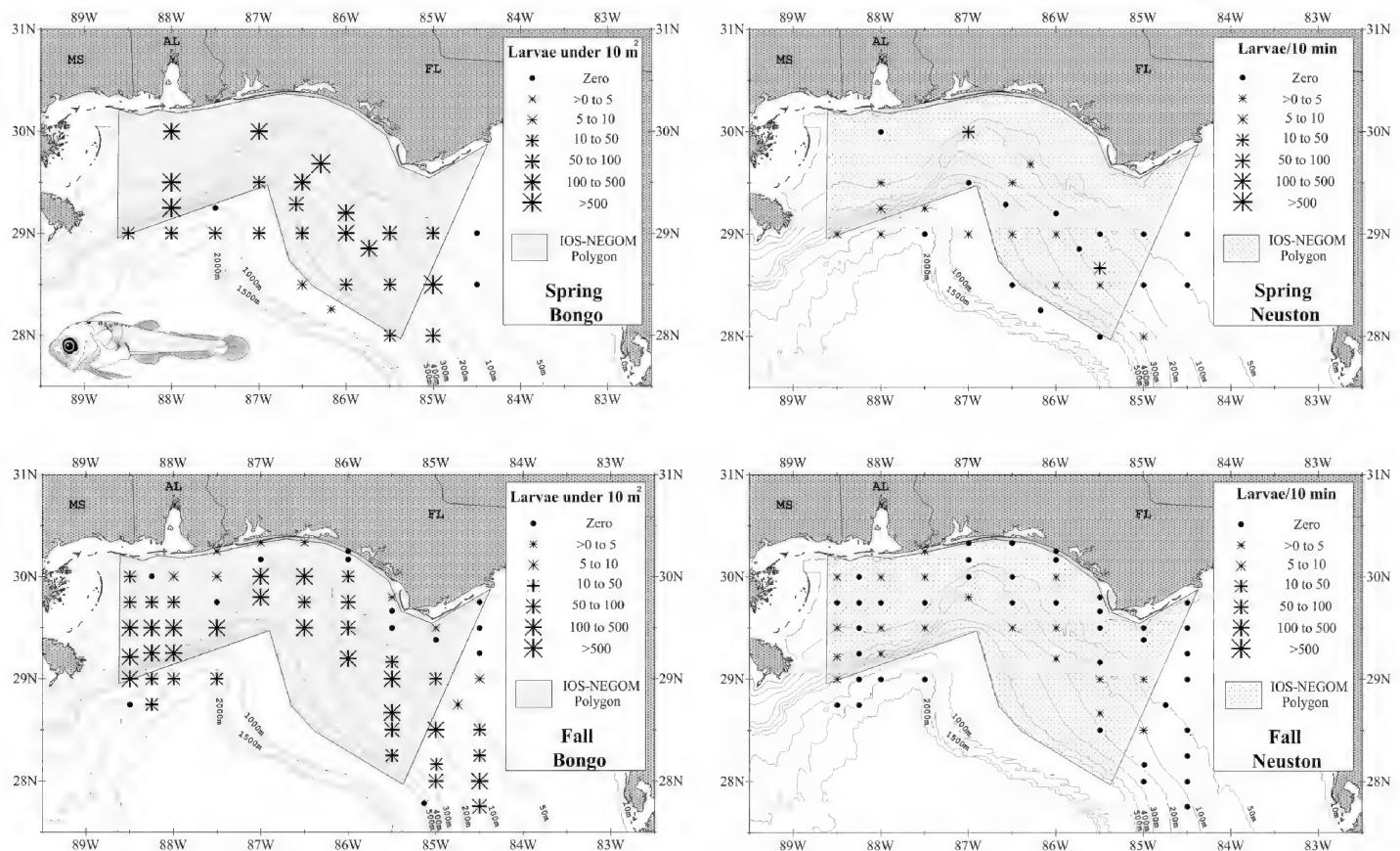
Figure 17. Mean abundance and mean CPUE of codlet (Bregmacerotidae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
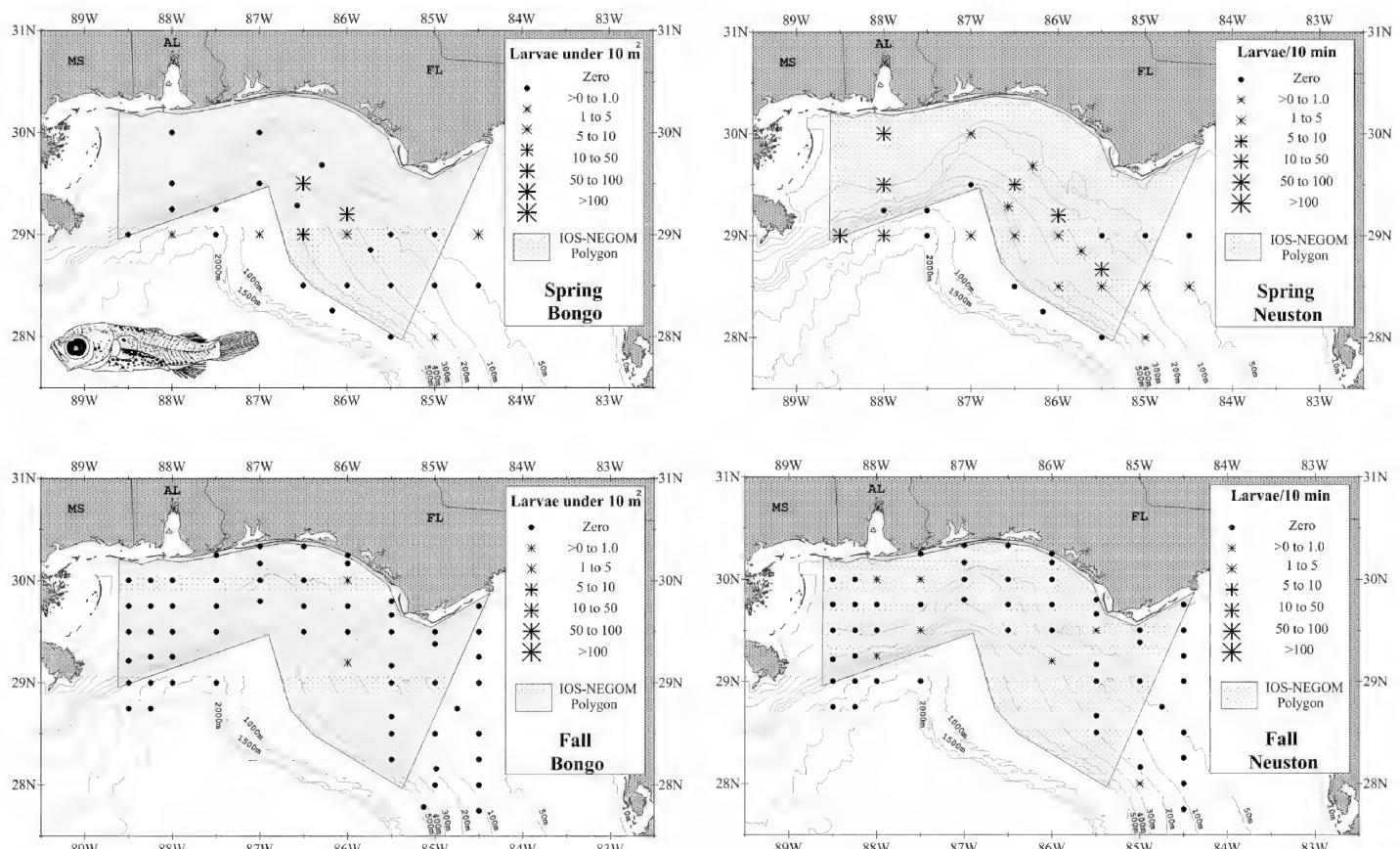
Figure 18. Mean abundance and mean CPUE of mullet (Mugilidae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
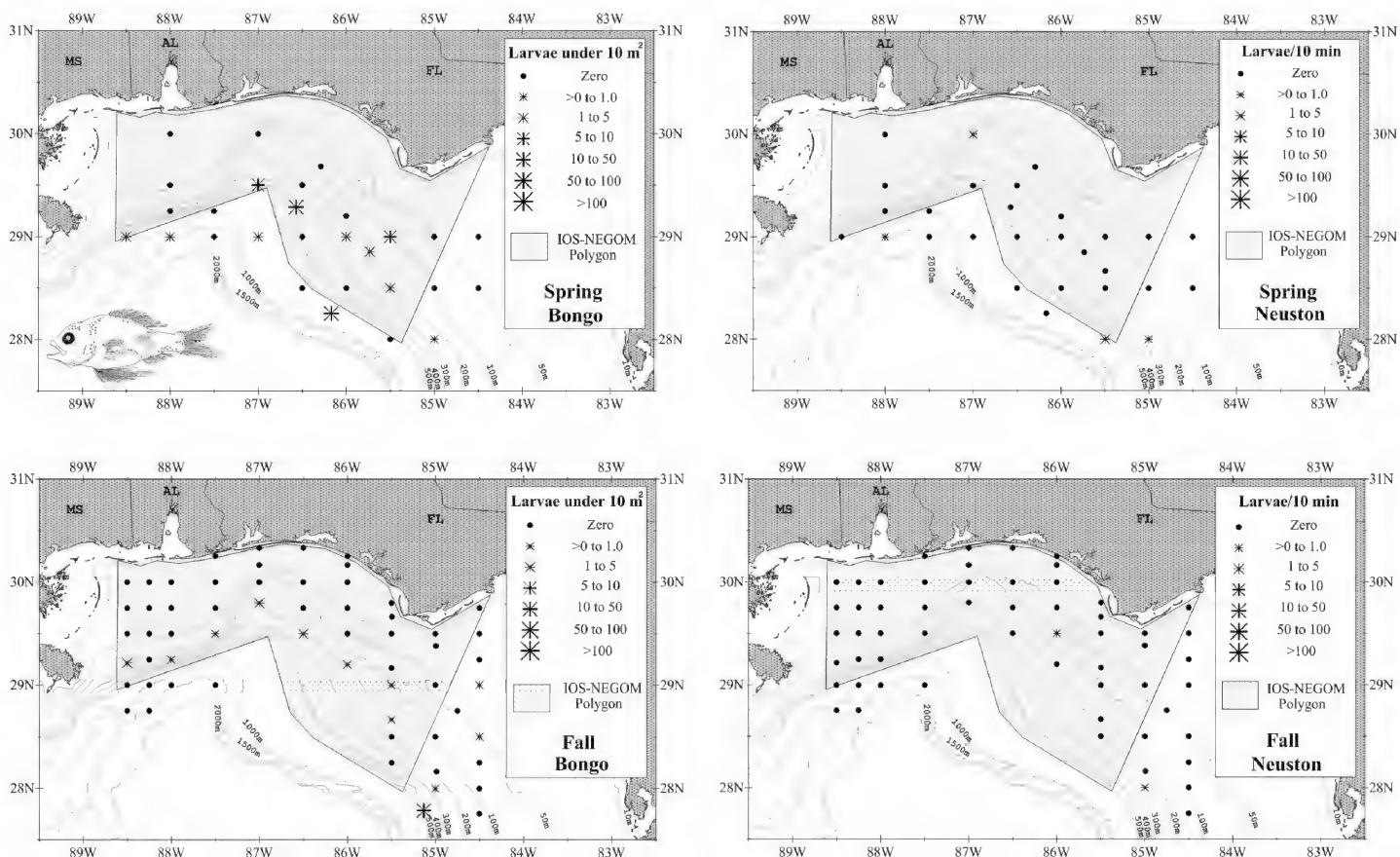
Figure 19. *Mean abundance and mean CPUE of bigscales (Melamphaidae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

spring than fall surveys (Table 1). The overall size range of melamphaid larvae captured in UNIS study area collections was 1.6–15.7 mm BL (n = 43). Most occurrences and highest mean abundances were observed at stations where station depths were ≥ 100 m (Figure 19). Bigscales or ridgehead larvae were taken significantly less often in study area bongo samples than Gulfwide during spring surveys (Table 3). Occurrence in spring neuston samples and during fall surveys was comparable in the 2 areas. Relative abundances and CPUEs in both areas were similar, differing by < 0.1% (Table 4).

HOLOCENTRIDAE (23 occurrences; 34 larvae; Figure 20)

The larvae and neustonic prejuvenile ('rhynchichthys') stage of these nocturnally active reef fishes are distinctive and unique among early life stages, yet identification beyond the family level is problematic (Lyczkowski–Shultz et al. 2000, Richards et al. 2006a). Squirrelfish larvae were taken primarily in neuston collections and were as frequently taken and as numerous in both spring and fall surveys (Table 1). Mean size in neuston collections was 10.4 mm BL (n = 18; range = 3.0–26.8 mm) and mean size in bongo samples was 2.0 mm BL (n = 4; range = 1.8–2.2 mm). Occurrences within the UNIS study area were restricted to stations where depths were ≥ 200 m (Figure 20). Most squirrelfish larvae, however, were taken at stations outside the IOS–NEGOM research

polygon; either at more offshore stations over depths > 500 m or to the southeast at comparable or shallower depths. Squirrelfish larvae were significantly less common in the study area than Gulfwide during both surveys (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.5% (Table 4).

SERRANIDAE (320 occurrences; 1,415 larvae)

Serranid larvae were nearly ubiquitous and homogeneously distributed throughout the UNIS study area and therefore, no distribution map is presented here. However, the distributions of 4 subfamilies did show differences and are presented below. Larvae were taken more frequently in bongo than neuston samples, during fall than spring surveys, and relatively more often in the study area than Gulfwide (Tables 1, 3). Mean size in bongo samples was 2.6 mm BL (n = 739; range = 1.2–11.0 mm) and mean size in neuston samples was 4.2 mm BL (n = 153; range = 2.7–11.5 mm). Larvae in this category were not identified beyond the family level because they had not developed certain key characteristics that would permit identification to one of the 5 serranid subfamilies. Larvae within these subfamilies are distinctive and can be identified once diagnostic characters such as head, dorsal and pelvic spines are developed (Richards 2006). It is likely that most of the larvae identified to the family level only belonged to the Subfamily Serraninae since larvae of this taxon
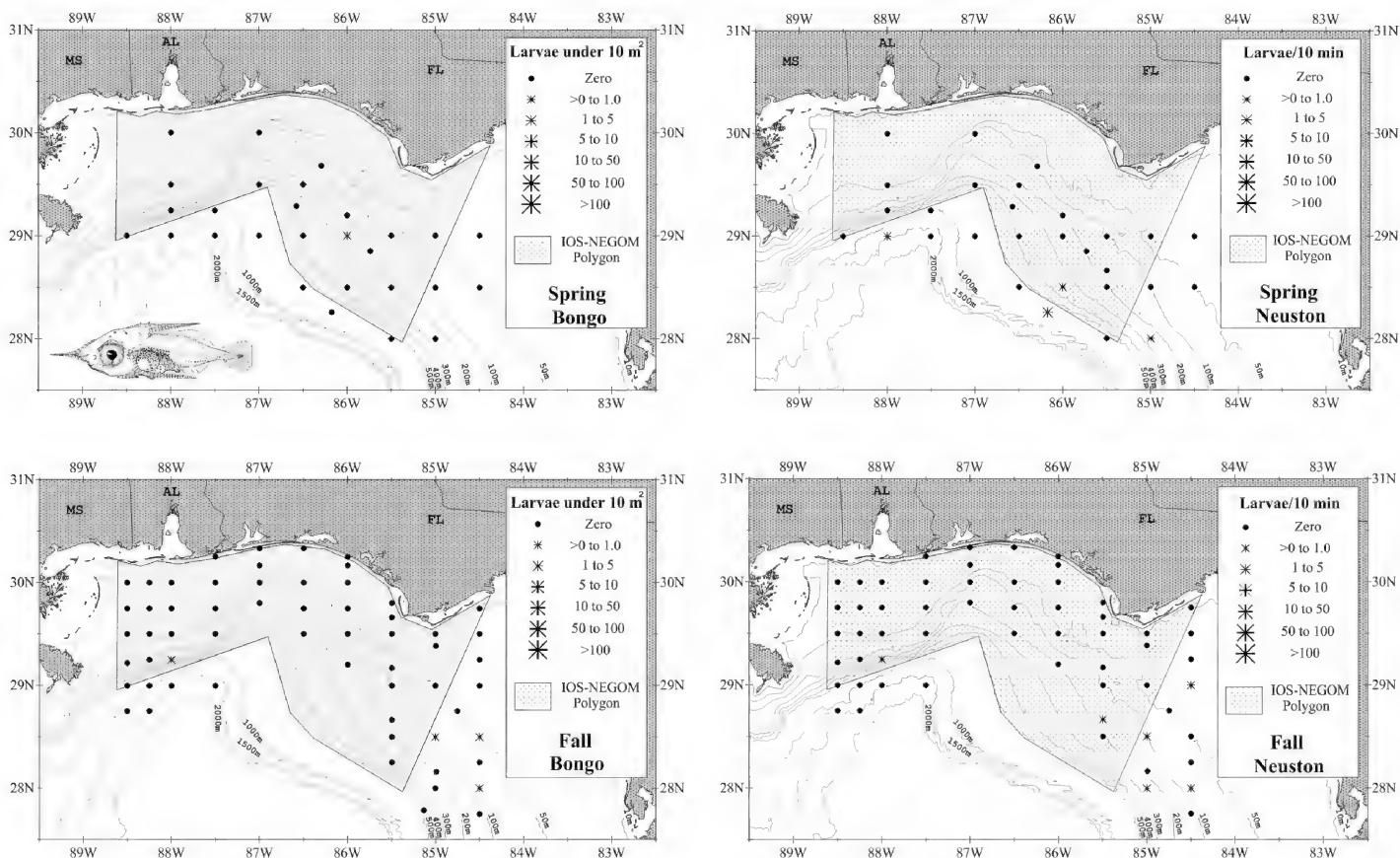
Figure 20. *Mean abundance and mean CPUE of squirrelfish (Holocentridae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*
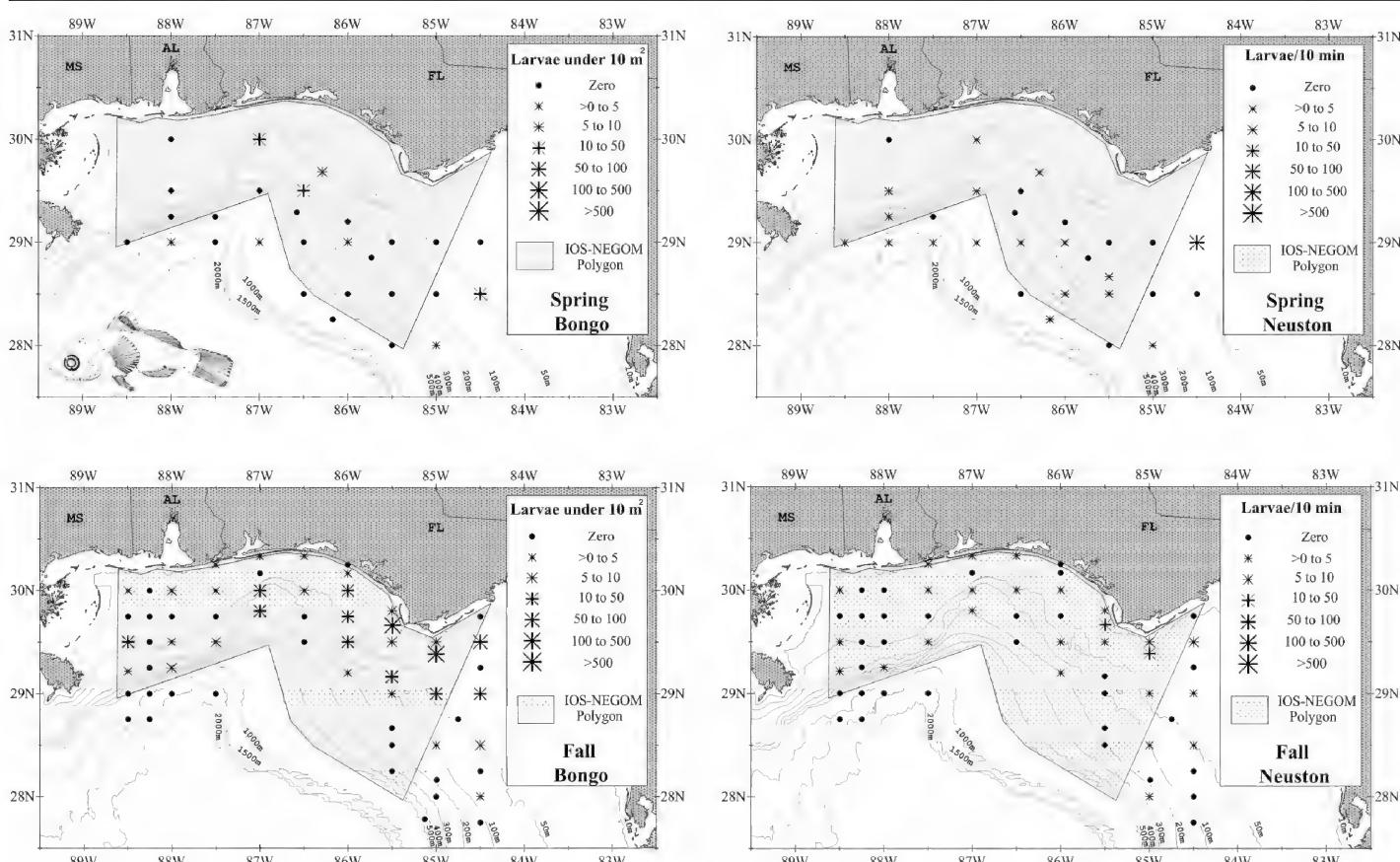

Figure 21. *Mean abundance and mean CPUE of Serraninae larvae (Family Serranidae) at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

66
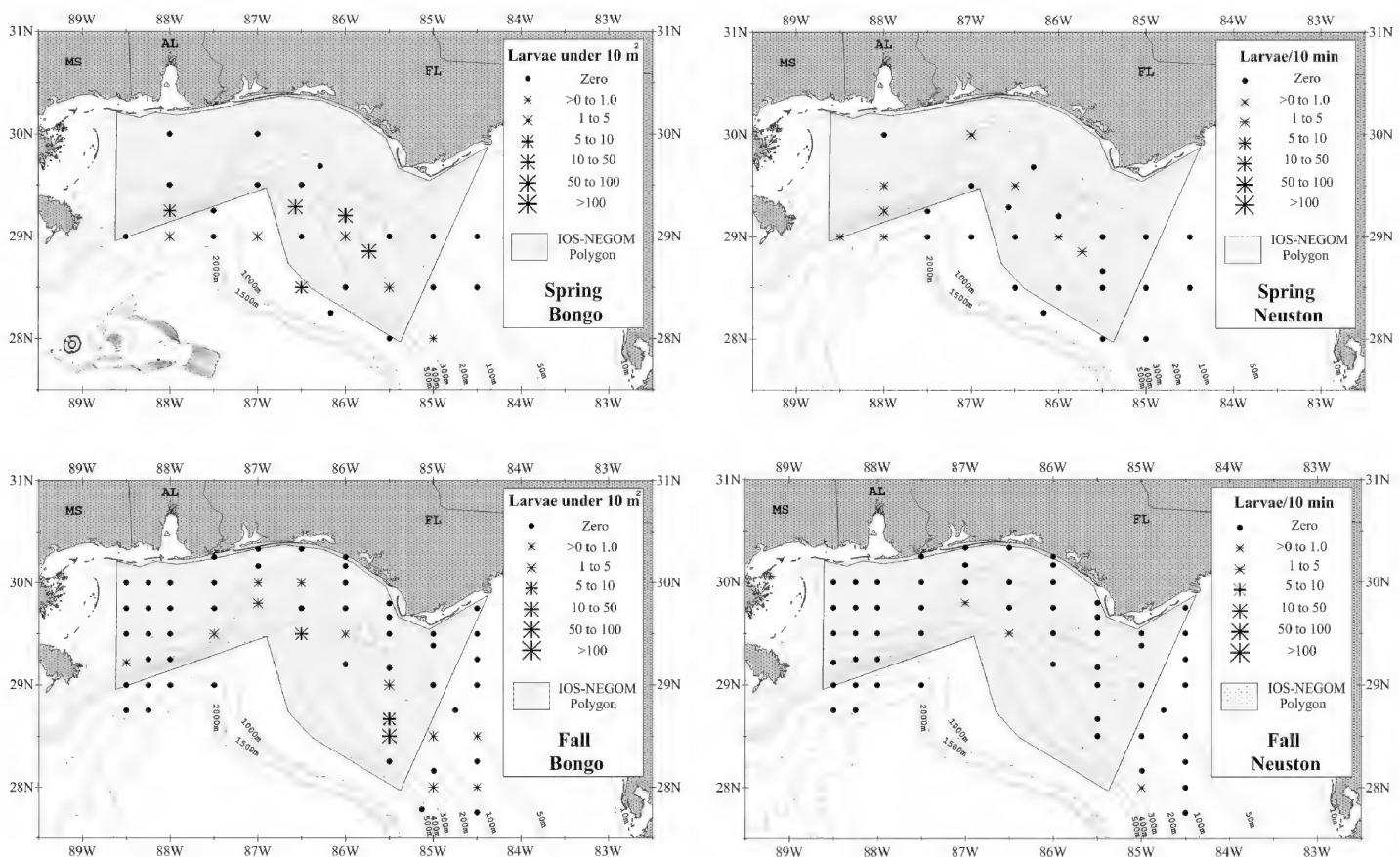
Figure 22. *Mean abundance and mean CPUE of Anthiinae larvae (Family Serranidae) at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

were the most numerous among larvae that could be identified to the subfamily level. Frequency of occurrence did not differ statistically between the UNIS and Gulfwide survey areas except in fall when serranid larvae were taken significantly more often in the study area than Gulfwide in bongo samples (Table 3). Relative abundances and CPUEs in the two areas differed by < 0.1% (Table 4).

SERRANINAE (236 occurrences; 1,672 larvae; Figure 21)

Sea bass larvae of the Subfamily Serraninae occurred most frequently and in greater numbers than the larvae of any of the other 4 subfamilies of sea basses. This taxon comprises species of genera such as the soft bottom dwelling *Centropristis* and *Diplectrum* and the reef dwelling *Hypoplectrus* and *Serranus*, among others. Over 67% of the occurrences and specimens of serranine larvae were captured in bongo samples and over 75% of the larvae were collected during fall surveys (Table 1). Mean size in neuston collections was 4.2 mm BL (n = 297; range = 2.1–11.5 mm) and mean size in bongo samples was 3.6 mm BL (n = 508; range = 1.5–12.1 mm). Larvae were found at both the shallowest and deepest stations during spring surveys, whereas during fall surveys, serranine larvae occurred most often at stations inshore of the 100 m isobath with highest mean abundances in the eastern region of the study area (Figure 21). Frequency of occurrence did not differ statistically between the UNIS and Gulfwide sur-

vey areas except in spring, when serranine larvae were taken significantly more often in the study area than Gulfwide in neuston samples (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.2% (Table 4).

ANTHIINAE (72 occurrences; 182 larvae; Figure 22)

Larvae of the Subfamily Anthiinae (streamer basses) were third in occurrence and abundance among sea bass larvae taken in the UNIS study area. In the GOM this subfamily comprises species in the genera, *Anthias, Hemanthias,* and *Pronotogrammus*. All species are abundant planktivores and ecologically important components of deep reef communities in the NEGOM (Weaver et al. 2002). Most larvae were taken in bongo samples and during spring surveys when the greatest mean abundances were found at stations between 200–400 m (Figure 22). Mean size in neuston collections was 3.9 mm BL (n = 55; range = 2.5–5.5 mm) and mean size in bongo samples was 3.1 mm BL (n = 108; range = 1.8–10.0 mm). Distribution of anthiine larvae during fall surveys closely followed the isobaths outlining the DeSoto Canyon between 87.5 and 85.5°W longitudes with larvae being collected at stations between 50–200 m (Figure 22). Frequency of occurrence did not differ statistically between the UNIS and Gulfwide survey areas except in fall when anthiine larvae were taken significantly more often in the study area than Gulfwide in bongo samples (Table 3). Relative abundances and
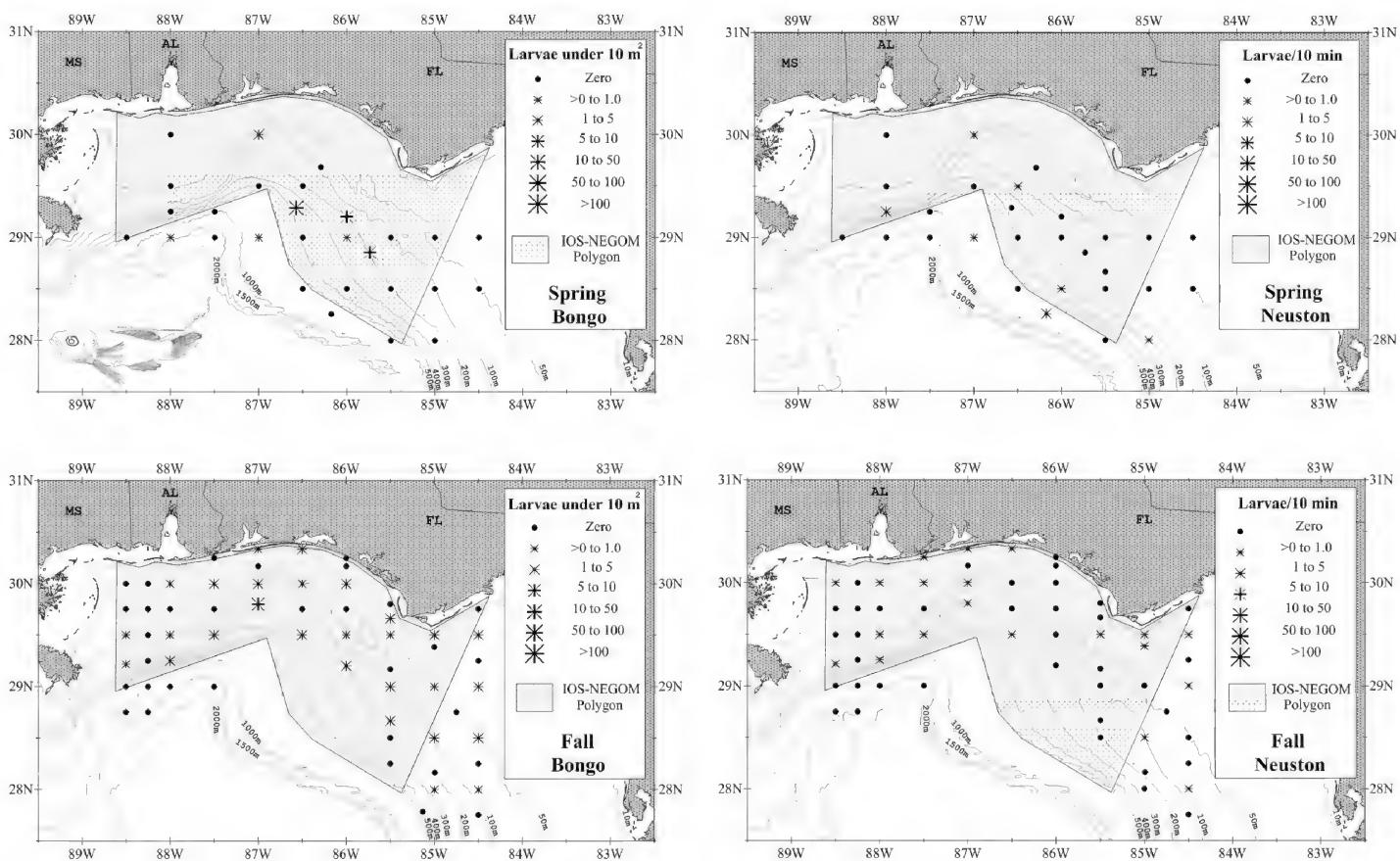
Figure 23. *Mean abundance and mean CPUE of Grammistinae larvae (Family Serranidae) at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

CPUEs in the 2 areas differed by < 0.3% (Table 4).

EPINEPHELINAE (3 occurrences; 3 larvae)

This subfamily includes the groupers, many of which are important resource species, and most of which are reef associates. A total of only 3 grouper larvae, ranging in size from 1.6–4.0 mm BL, were taken; all in bongo samples and all during spring surveys (Table 1). Larvae were taken at 3 stations within the IOS–NEGOM research polygon located on the 200 m isobath between 86 and 85.5°W longitudes and at a third stations outside the IOS–NEGOM research polygon along the 87°W meridian between 500–1000 m. Grouper larvae were captured an additional 32 times Gulfwide: in spring (27 occurrences) and fall (5 occurrences); and in bongo (30 occurrences) and neuston (2 occurrences) samples (Table 3). A recent re–examination of all grouper larvae (n = 474) collected Gulfwide during SEAMAP surveys (1982–2005) now provides a more comprehensive description of the abundance and distribution of grouper larvae in the GOM (Marancik et. al. 2010, 2012); therefore, no distribution map is presented here.

GRAMMISTINAE (117 occurrences; 215 larvae; Figure 23)

Larvae of the soapfish subfamily were the second most common among serranid larvae in the UNIS study area. Larvae were most often captured in bongo than neuston samples, and over 80% were taken during fall surveys. Mean size in neuston collections was 5.4 mm BL (n = 61; range =

2.8–16.0 mm) and mean size in bongo samples was 3.9 mm BL (n = 125; range = 1.8–11.8 mm). Soapfish larvae were distributed throughout the study area but were more commonly found east of longitude 87.5°W (Figure 23). Frequencies of occurrence in the UNIS and Gulfwide survey areas were not statistically different during either survey (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

PRIACANTHIDAE (109 occurrences; 239 larvae; Figure 24)

Priacanthids (bigeyes) are shallow to deep water, hard–bottom dwelling fishes, and typical inhabitants of NEGOM deep reefs (Weaver et. al. 2002). Their life history includes a pelagic juvenile stage (Watson 1996a). Larvae were equally represented in bongo and neuston collections in the UNIS study area; however, most occurrences and specimens were taken during fall surveys (Table 3). Mean size in neuston collections was 4.7 mm BL (n = 49; range = 2.4–18.0 mm) and mean size in bongo samples was 2.9 mm BL (n = 71; range = 1.4–6.8 mm). Priacanthid larvae were distributed throughout the study area but larvae were taken more often at stations east of 87°W longitude (Figure 24). Frequencies of occurrence in the UNIS and Gulfwide survey areas were not statistically different and relative abundances and CPUEs in the 2 areas differed by < 0.1% (Tables 3, 4).

APOGONIDAE (169 occurrences; 579 larvae; Figure 25)

The cardinalfishes are planktivorous, nocturnal fishes
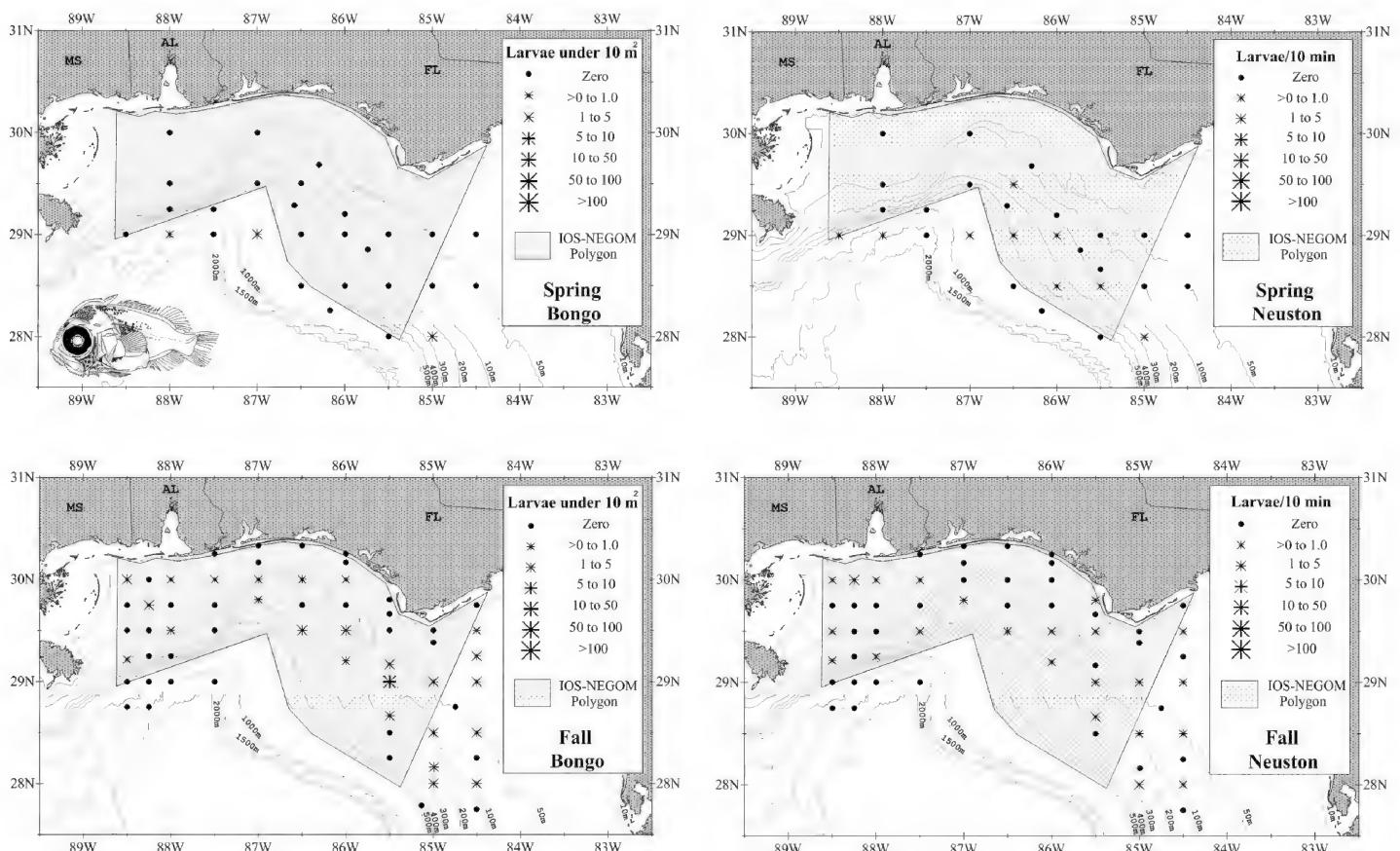
**Figure 24.** Mean abundance and mean CPUE of bigeye (Priacanthidae) larvae at stations in the UNIS study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
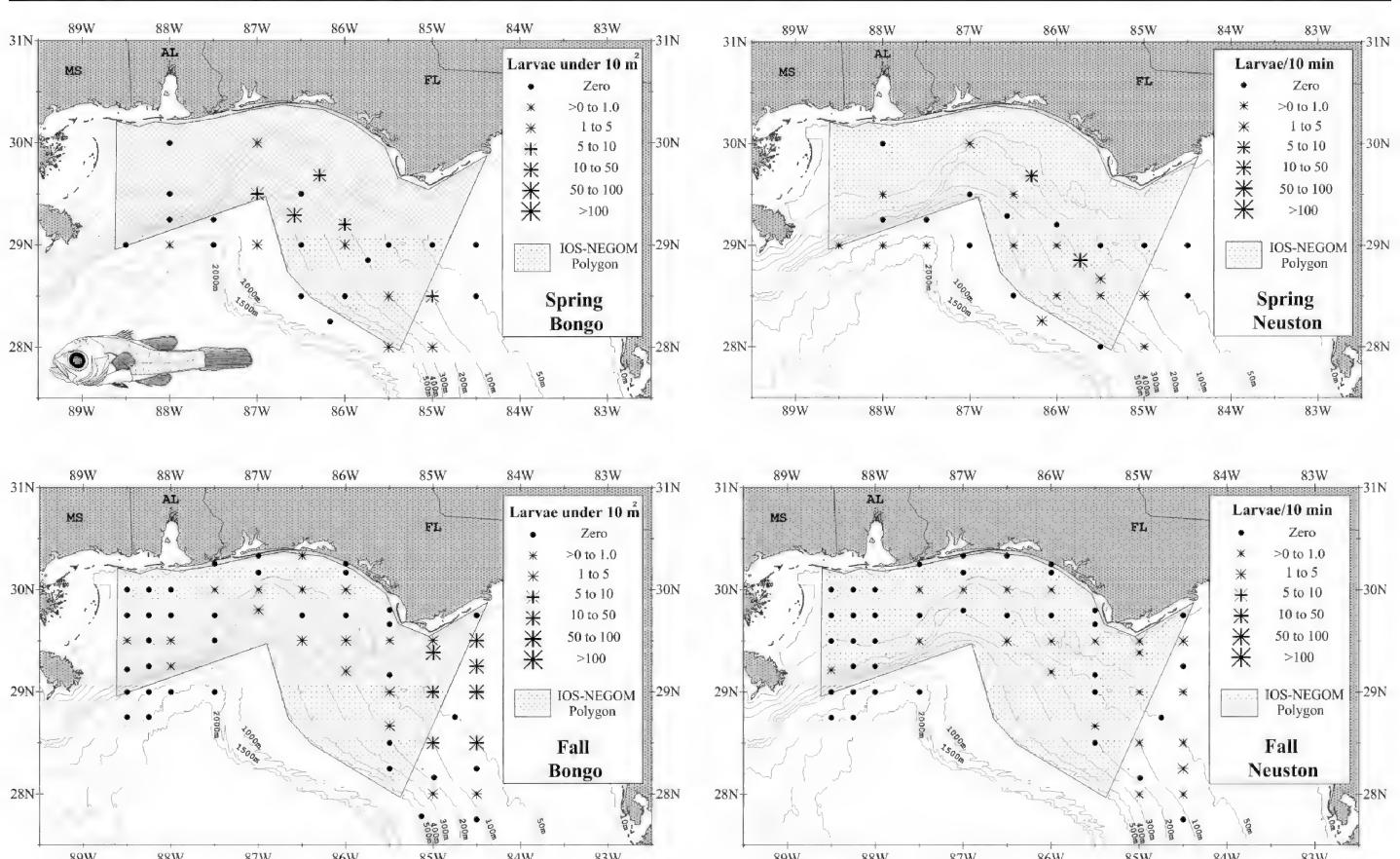
Figure 25. Mean abundance and mean CPUE of cardinalfish (Apogonidae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.

usually associated with reefs whose larvae hatch with functional mouths and pigmented eyes; subsequently, the larvae of many species are mouth–brooded prior to dispersal into the plankton (Thresher 1984). Definitive identification of larvae as belonging to the family Apogonidae is problematic prior to median fin base formation (Leis and Rennis 1983). Despite this uncertainty, data on SEAMAP collected larvae identified as apogonids are summarized herein. Potentially misidentified larvae (i.e., belonging to another fish family) are likely an insignificant fraction of the total putative apogonids. Cardinalfish larvae were captured only slightly more often in bongo than in neuston samples but larvae were more common and numerous in fall than in spring survey samples (Table 1). Mean size in neuston collections was 4.5 mm BL (n = 60; range = 2.9–15.0 mm) and mean size in bongo samples was 3.6 mm BL (n = 94; range = 1.5–10.5 mm). Apogonid larvae were taken more often and were more numerous at stations east of 87°W longitude during both surveys (Figure 25). Larvae were significantly more common in the UNIS than Gulfwide survey area during spring surveys in neuston samples but were significantly less common in the UNIS study area during fall surveys in bongo samples (Table 3). Frequencies of occurrence in spring bongo and fall neuston samples were not significantly different in the 2 areas. Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

RACHYCENTRIDAE

Rachycentron canadum (5 occurrences; 21 larvae; Figure 26)

The larvae of this highly prized recreational and coastal migratory species are rarely taken in plankton collections. Cobia larvae occurred only 5 times in UNIS study area collections, all in neuston samples and most specimens (17) during spring surveys (Table 1). Mean size of larvae was 10.9 mm BL (n = 7; range = 7.0–21 mm). Larvae were captured at 4 different stations, all along or west of 87°W longitude (Figure 26). Water depth at the stations of capture during spring surveys was ≥ 500 m, whereas water depth at capture

stations during fall surveys was ~200 m. More cobia larvae were taken in the study area during spring than fall surveys while the reverse was true Gulfwide. Frequency of occurrence of cobia larvae was not significantly different between the study area and Gulfwide during spring surveys (Table 3). However, despite the few captures overall, occurrence in neuston samples was significantly less in study area samples during fall surveys than Gulfwide. Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

CORYPHAENIDAE (187 occurrences; 438 larvae; Figure 27)

The young of two species of dolphins are combined in this taxon, *Coryphaena equisetis* and *C. hippurus*. Although taken in bongo samples as well, most *Coryphaena* larvae were collected in neuston samples, with over half the occurrences and over 60% of the specimens being taken during spring surveys (Table 1). The overall size range of young dolphin captured in UNIS study area collections was 3.0–31 mm BL (n = 14) in bongo samples and 3.0–105 mm BL (n = 251) in neuston samples. Young dolphins were distributed throughout the UNIS study area during both survey timeframes. Mean abundances were fairly uniform across the study area in the spring but higher mean abundances were observed in the western region during fall surveys (Figure 27). The frequency of occurrence of dolphin larvae in spring bongo samples and during fall surveys was not significantly different between the study area and Gulfwide (Table 3), but occurrence in spring survey neuston samples was significantly less in the study area than Gulfwide. Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

CARANGIDAE

Caranx spp. (183 occurrences; 1,449 larvae; Figure 28)

The young of 6 species of jacks, all important forage fish, cannot be reliably separated, although identification to genus even at small sizes is straightforward. Larvae were taken primarily in neuston samples but with equal frequency in both spring and fall surveys (Table 1). The overall size range of jack larvae captured in UNIS study area collections was
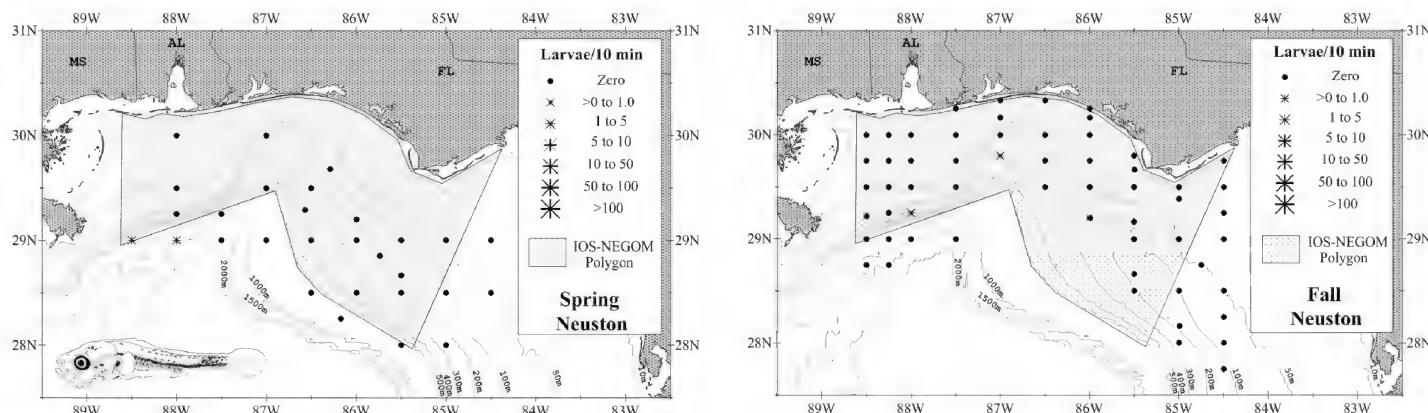
Figure 26. Mean CPUE of cobia, Rachycentron canadum, larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
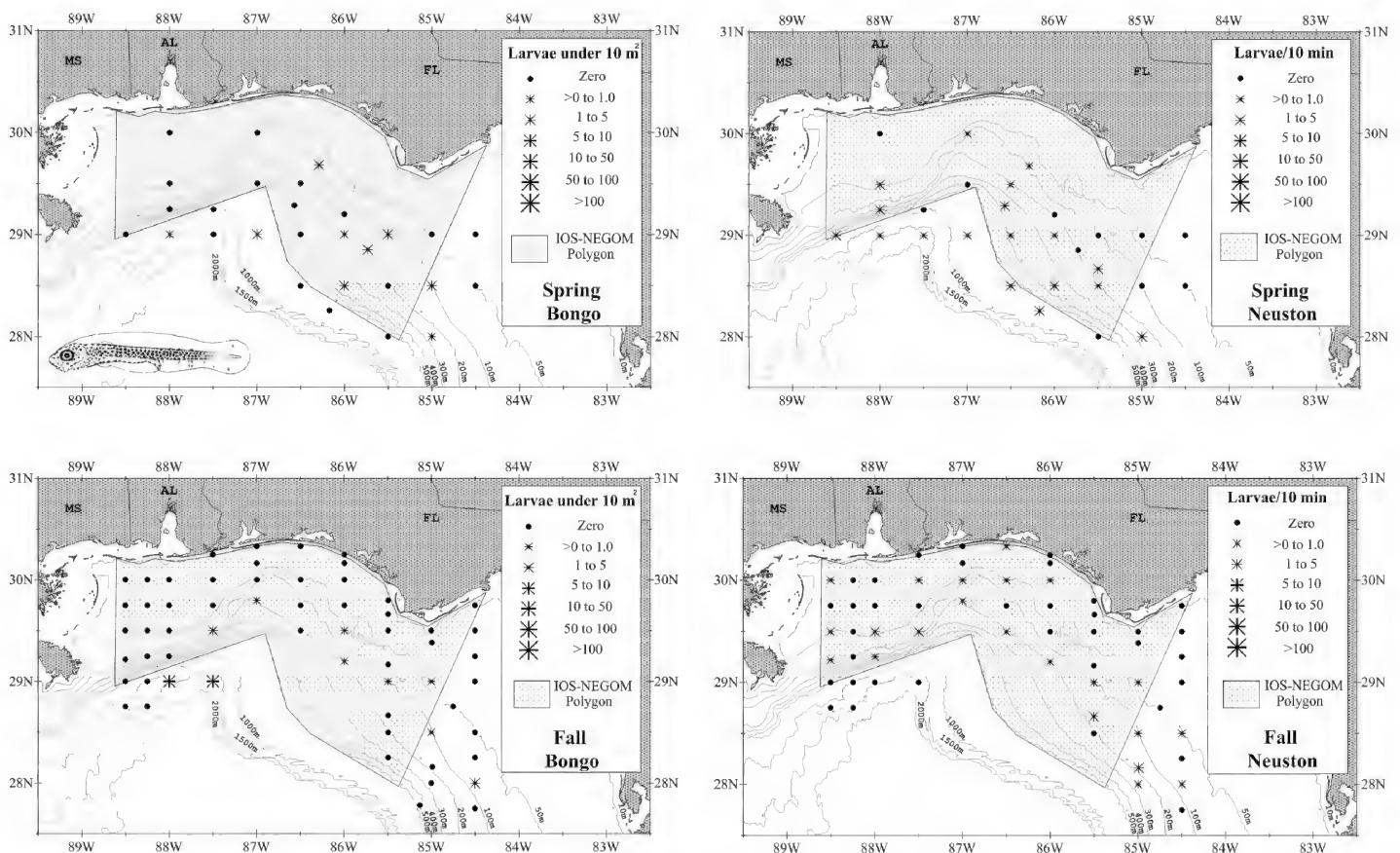
Figure 27. Mean abundance and mean CPUE of dolphin (Coryphaenidae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
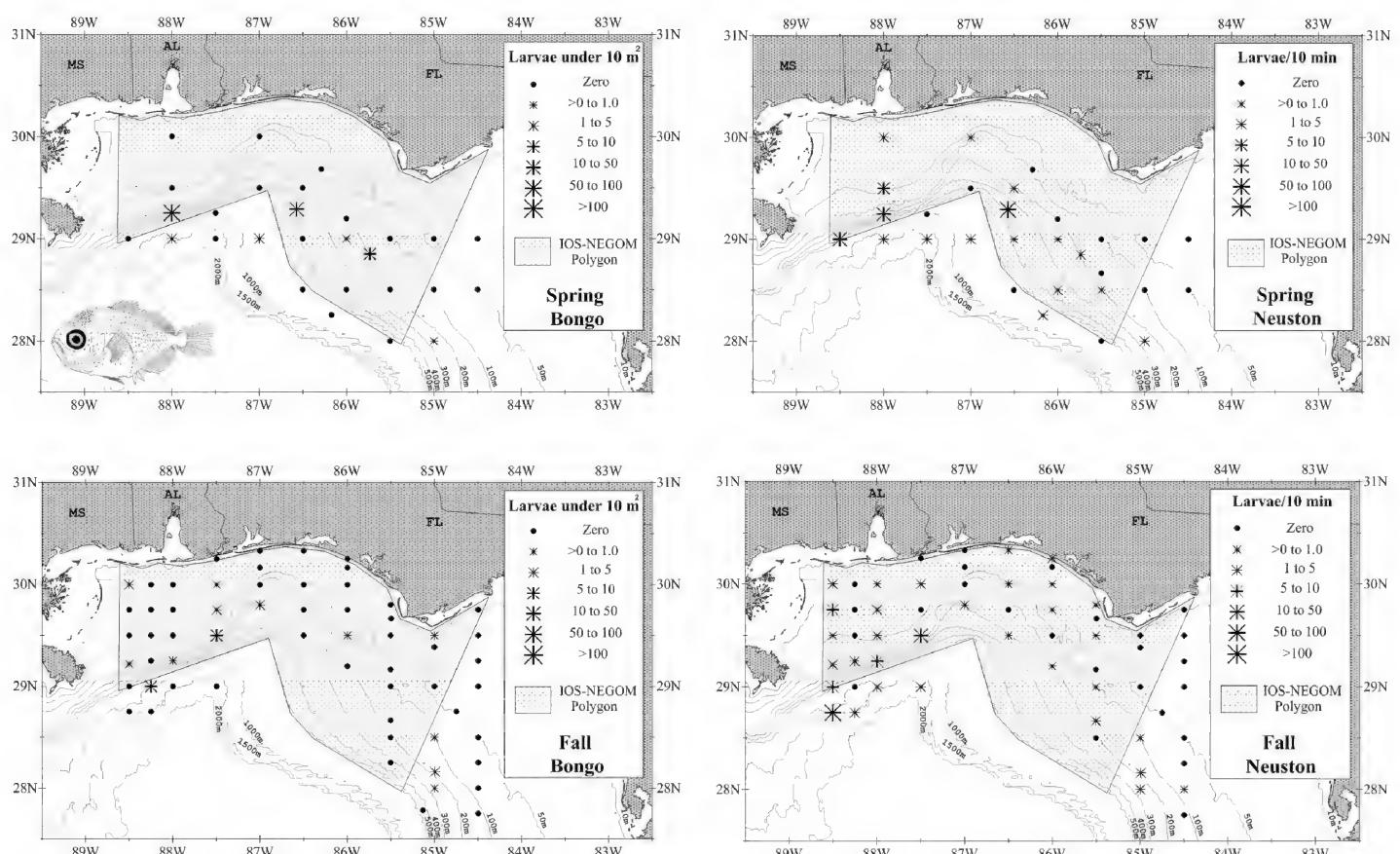

Figure 28. Mean abundance and mean CPUE of jack larvae, genus Caranx, at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.

2.2–55 mm BL. Jack larvae occurred more consistently and in greater numbers in the western region of the IOS–NE-GOM research polygon (west of 87°W longitude) then in the eastern region (Figure 28).

Jack larvae were significantly less common in study area spring neuston samples and during fall surveys than Gulf-wide (Table 3). Frequency of occurrence in spring bongo samples was not significantly different between the 2 areas. Relative abundances and CPUEs in the 2 areas differed by < 4% (Table 4).

Chloroscombrus chrysurus (206 occurrences; 14,916 larvae; Figure 29)

Throughout the species range, Atlantic bumper young are most often seen in commensal association with jellyfishes. Larvae and juveniles were second in occurrence but first in abundance among carangid larvae captured in the UNIS study area. Although they were taken almost as frequently in bongo as in neuston samples, over 90% of specimens were captured in neuston collections and all but one occurrence (8 specimens) came during fall surveys (Table 1). Mean size in bongo samples was 3.2 mm BL (n = 281; range = 1.2–40 mm) and mean size in neuston samples was 7.3 mm BL (n = 252; range = 1.8–42 mm). Larvae were taken throughout the study area, but the highest mean abundances were well within the 50 m isobath and at the northernmost stations of the study area (Figure 29). The single occurrence of a larvae in spring (not shown in Figure 29) was at station B176 over 50 m water depth. Atlantic bumper, like *Caranx* spp., larvae were taken more consistently and in greater numbers farther offshore in the southwestern than the southeastern region of the study area. The occurrence of Atlantic bumper larvae in spring neuston samples was comparable in both areas but larvae were significantly less common in the study area than Gulfwide in fall bongo and neuston samples (Table 3). Relative abundance and CPUE in the 2 areas differed by < 0.1% in spring survey samples and by < 7% in fall survey samples (Table 4).

Decapterus spp. (479 occurrences; 7,101 larvae; Figure 30)

Three species of *Decapterus* may occur in the GOM but the most common one is the round scad, *D. punctatus*. Therefore, it is likely that the majority of larvae identified to this taxon are *D. punctatus* larvae. *Decapterus* larvae were the most frequently captured and second most abundant among carangid larvae in the UNIS study area. These larvae were as frequently captured in bongo as in neuston samples but, unlike Atlantic bumper larvae, they were also as numerous in bongo as in neuston samples (Table 1). Most larvae, 87% of occurrences and 94% of specimens, were taken during fall surveys. Mean size in bongo samples was 3.2 mm BL (n = 1002; range = 1.2–61 mm) and mean size in neuston samples was 8.5 mm BL (n = 866; range = 1.5–57 mm). Unlike the 2 previous carangid taxa, *Decapterus* larvae were nearly homogeneously distributed throughout the UNIS study area from east to west and onshore to offshore (Figure 30). *Decapterus* larvae also differed from *Caranx* and *Chloroscombrus* larvae in that they occurred more frequently in the study area than Gulfwide in all but one survey/gear combination (Table 3). The frequency of occurrence of *Decapterus* larvae in spring neuston samples and fall bongo and neuston samples was significantly higher in the study area than Gulfwide (Table 3). Occurrence in spring bongo samples was comparable in the 2 areas. Relative abundances and CPUEs in the 2 areas differed by < 3% (Table 4).

Selar crumenophthalmus (99 occurrences; 710 larvae; Figure 31)

Young bigeye scad, although relatively numerous, did not occur as frequently as the previously treated carangid taxa. Bigeye scad larvae were captured about as often in bongo and neuston collections but most specimens (83%) were taken in neuston samples during fall surveys (Table 1). Mean size in bongo samples was 2.7 mm BL (n = 81; range = 1.4–7.0 mm) and mean size in neuston samples was 4.3 mm BL (n = 83; range = 2.8–11.0 mm). Larvae were widely distributed from east to west within the IOS–NEGOM research polygon but
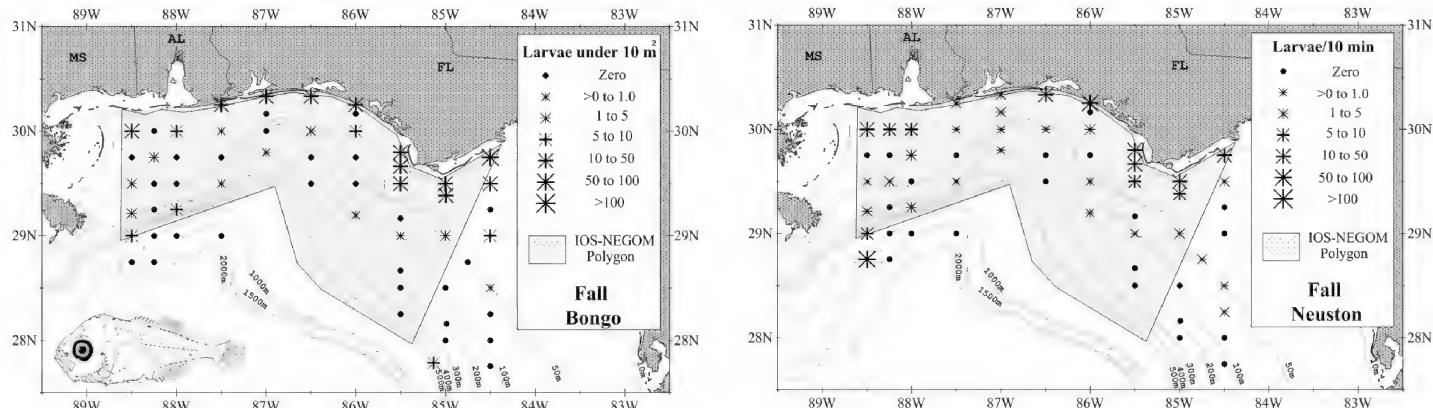
Figure 29. *Mean abundance and mean CPUE of Atlantic bumper,* Chloroscombrus chrysurus, *larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. (Map of one occurrence in spring survey neuston samples is not shown.) UNIS and SEAMAP defined in Figure 1.*
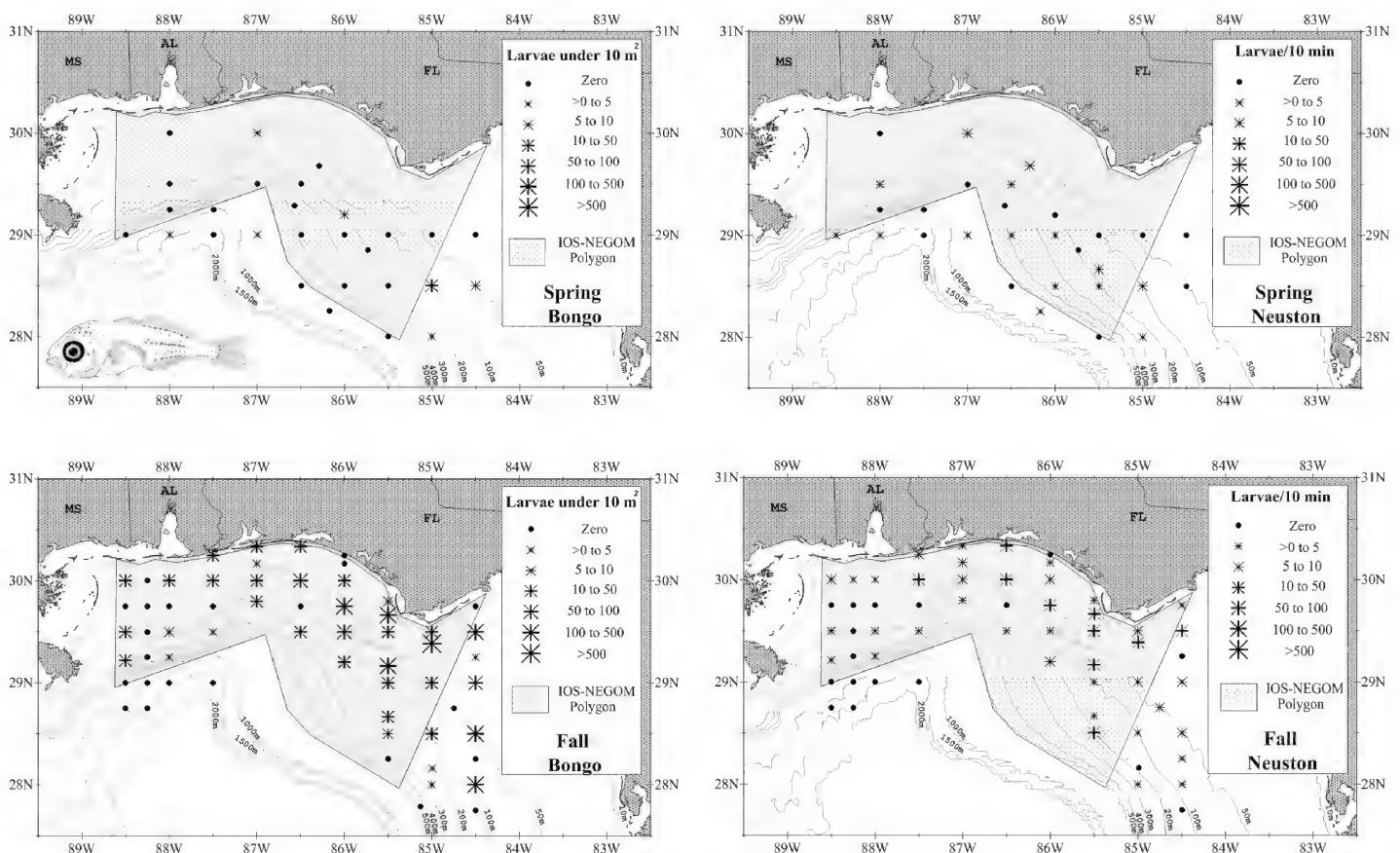
Figure 30. Mean abundance and mean CPUE of scad larvae, genus Decapterus, at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
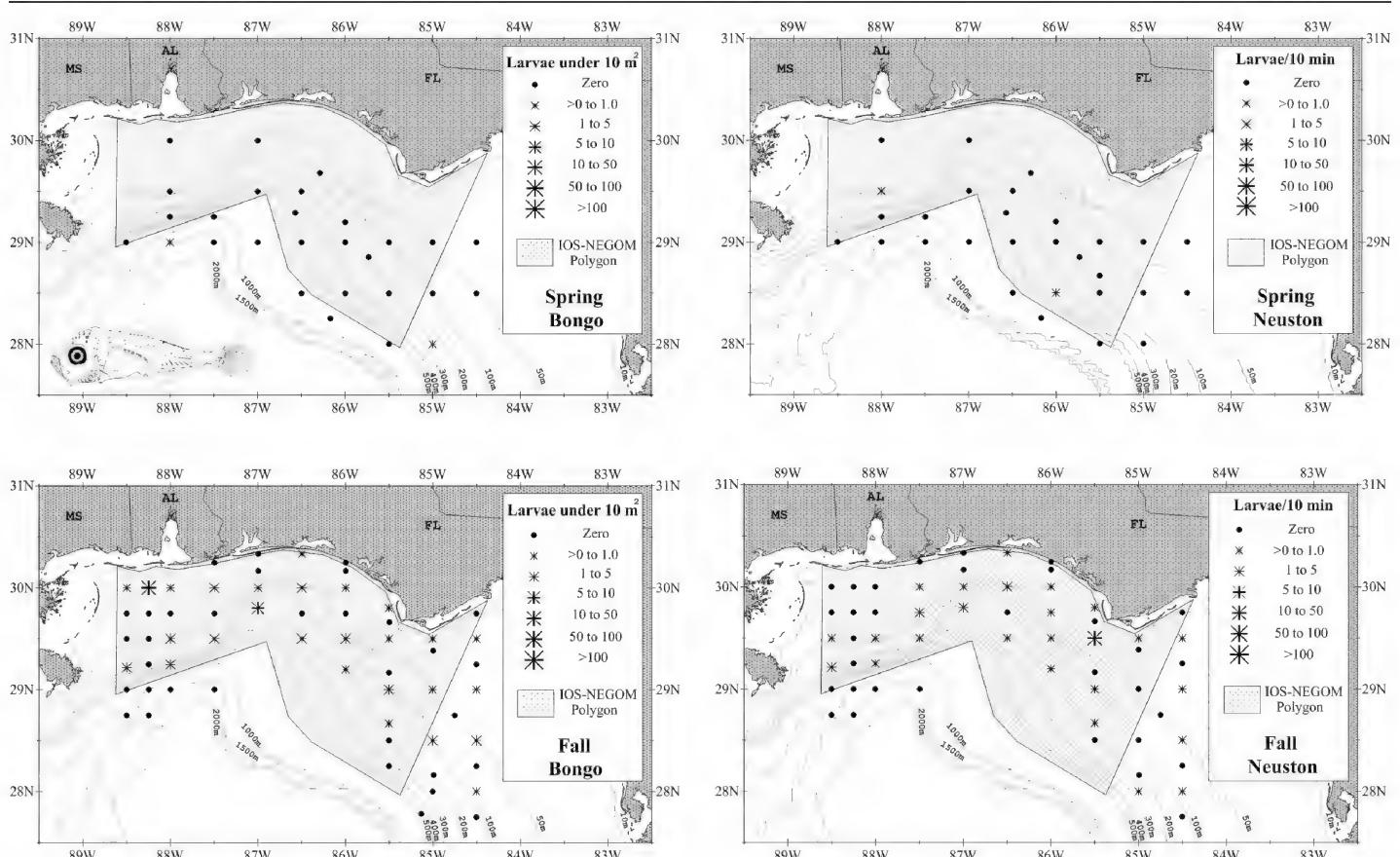

Figure 31. Mean abundance and mean CPUE of bigeye scad, Selar crumenophthalmus, larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.

were more restricted in onshore/offshore distribution, with most occurrences at stations between the 50–200 m isobaths (Figure 31). Frequency of occurrence of bigeye scad was comparable in the 2 areas in all instances except for spring neuston samples, when occurrence was significantly less in the study area than Gulfwide (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.7% (Table 4).

Selene spp. (34 occurrences; 53 larvae; Figure 32)

The young of 3 species of moonfish and lookdowns may occur in the GOM. Larvae were taken as often and in about the same numbers in bongo and neuston samples, but all captures were made during fall surveys (Table 1). Larvae ranged in overall size from 2.0–20.2 mm. Mean length in bongo and neuston samples was 3.7 mm (n = 24) and 5.3 mm (n = 18), respectively. Most occurrences were at stations between 50–200 m, but captures over greater depths were made in the southwestern region of the UNIS study area (Figure 32). *Selene* larvae were taken significantly less frequently in UNIS area bongo samples than Gulfwide, but occurrence in neuston samples was comparable in the 2 areas (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).
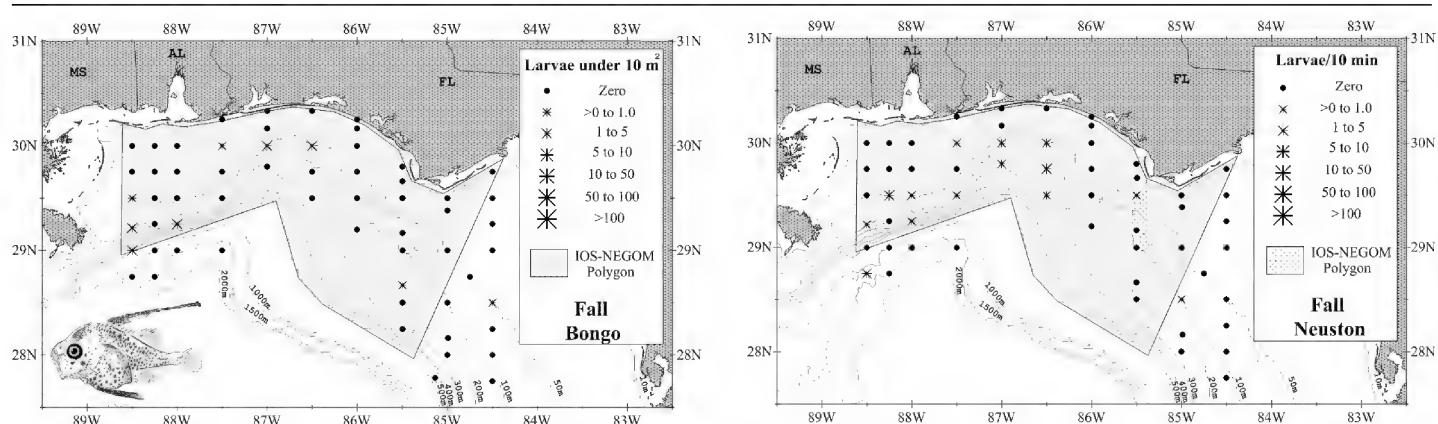
Figure 32. *Mean abundance and mean CPUE of moonfish and lookdown larvae, genus* Selene, *at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*
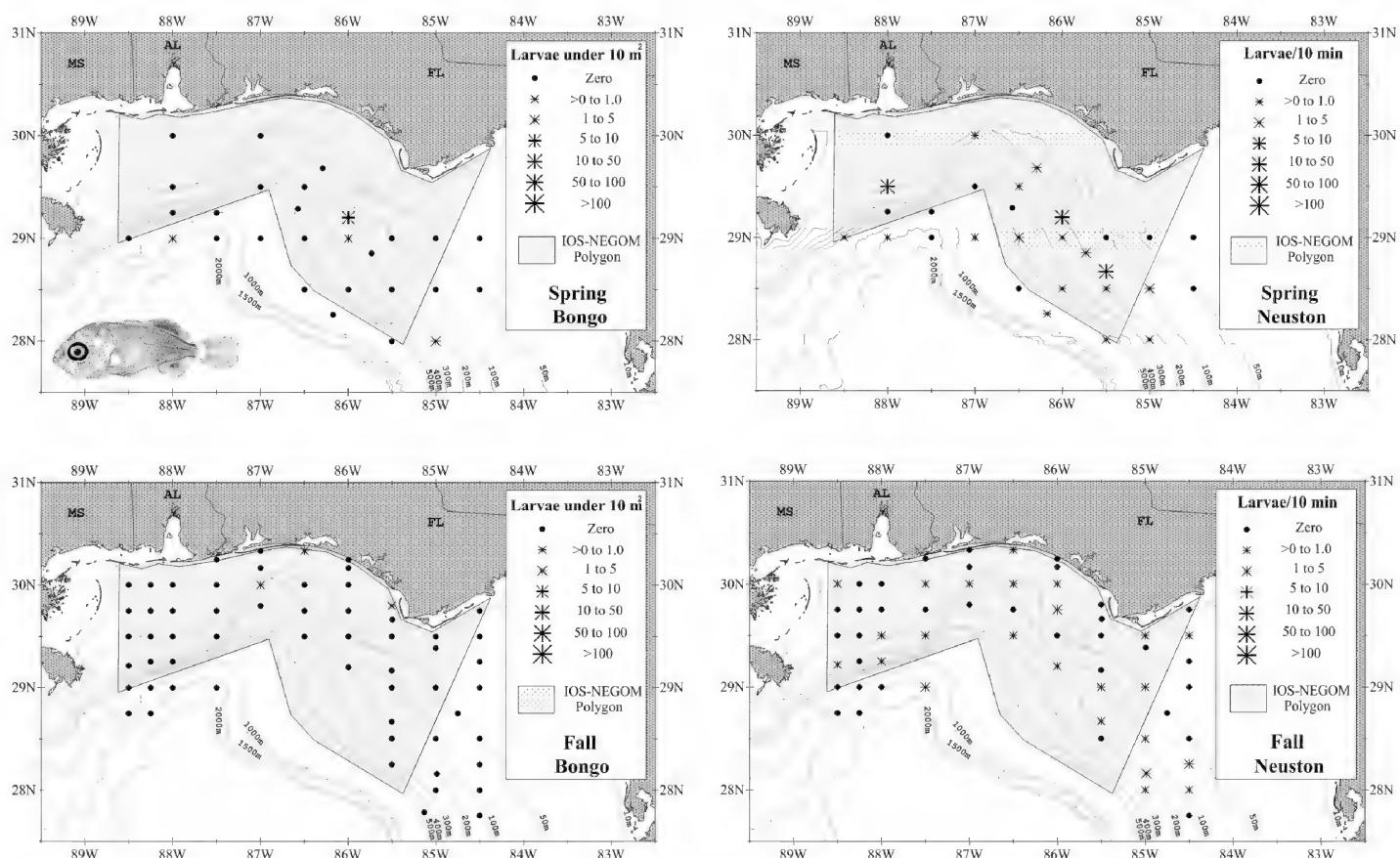
Figure 33. *Mean abundance and mean CPUE of amberjack larvae, genus* Seriola, *at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

Seriola spp. (123 occurrences; 461 larvae; Figure 33)

Four species of amberjacks may be represented among specimens in this taxon since species identification among *Seriola* larvae remains problematic despite a few incomplete larval descriptions (Laroche et al. 2006). Young amberjack were taken almost exclusively in neuston collections and during spring surveys (Table 1). Amberjack larvae ranged in size from 2.7–44 mm in UNIS study area collections. Mean length in neuston samples was 8.2 mm (n = 186). Although amberjack larvae were taken throughout the study area most captures were made at stations east of 87°W longitude (Figure 33). *Seriola* larvae were captured significantly more often in spring neuston samples but less often in fall bongo samples in the study area than Gulfwide (Table 3). Frequency of capture in spring bongo and fall neuston samples was comparable in the 2 areas. Relative abundances and CPUEs in the 2 areas differed by < 0.2% (Table 4).

Trachinotus spp. (46 occurrences; 85 larvae; Figure 34)

Three or 4 species of pompanos may be represented among the larvae and juveniles in this taxon, all of which were captured in neuston samples and mostly during spring surveys (Table 1). Mean size of young pompano was 7.1 mm BL (n = 61; range = 4.5–11.4 mm). Pompano young occurred not only at the shallowest, nearshore stations but also at some of the furthest offshore stations. This pattern may be indicative of species–specific distribution patterns among the larvae of the different *Trachinotus* species included in this taxon (Figure 34). Pompano larvae were taken statistically more often in spring neuston samples in the UNIS than Gulfwide survey area, but during fall surveys frequency of occurrence was comparable in the 2 survey areas (Table 3). Relative abundances and CPUEs in the 2 areas differed by <0. 1% (Table 4).

Trachurus lathami (16 occurrences; 61 larvae; Figure 35)

The young of this late winter spawning species were taken mostly in neuston samples and only during spring surveys (Table 1). Mean size in bongo samples was 4.1 mm BL (n = 7; range = 3.0–4.6 mm) and mean size in neuston samples was 4.5 mm BL (n = 28; range = 3.0–6.3 mm). Most rough scad larvae were captured at stations near or beyond the 200 m isobath across the NEGOM research polygon (Figure 35). Frequencies of occurrence in the UNIS and Gulfwide survey
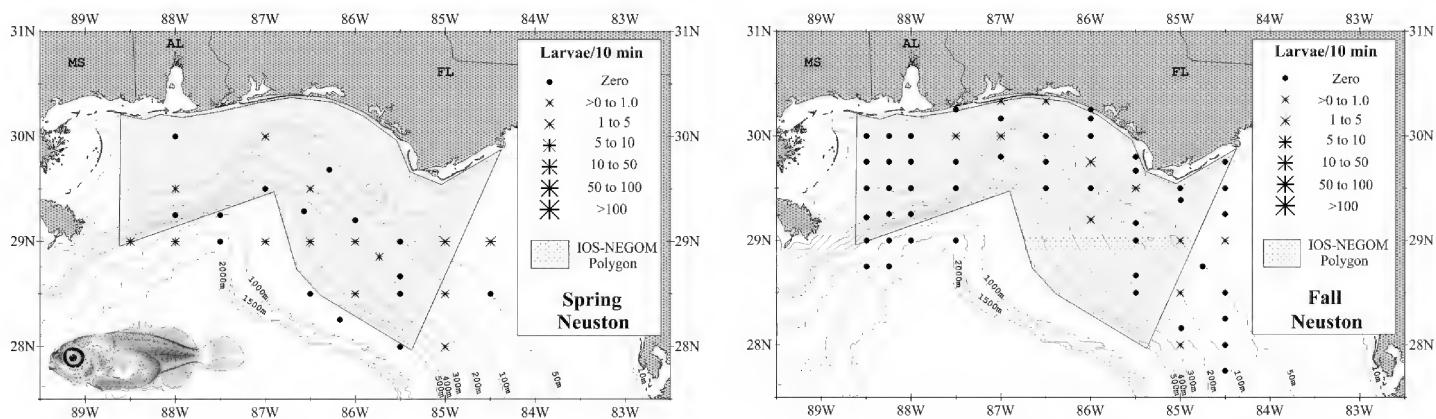
Figure 34. *Mean CPUE of pompano larvae, genus* Trachinotus, *at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*
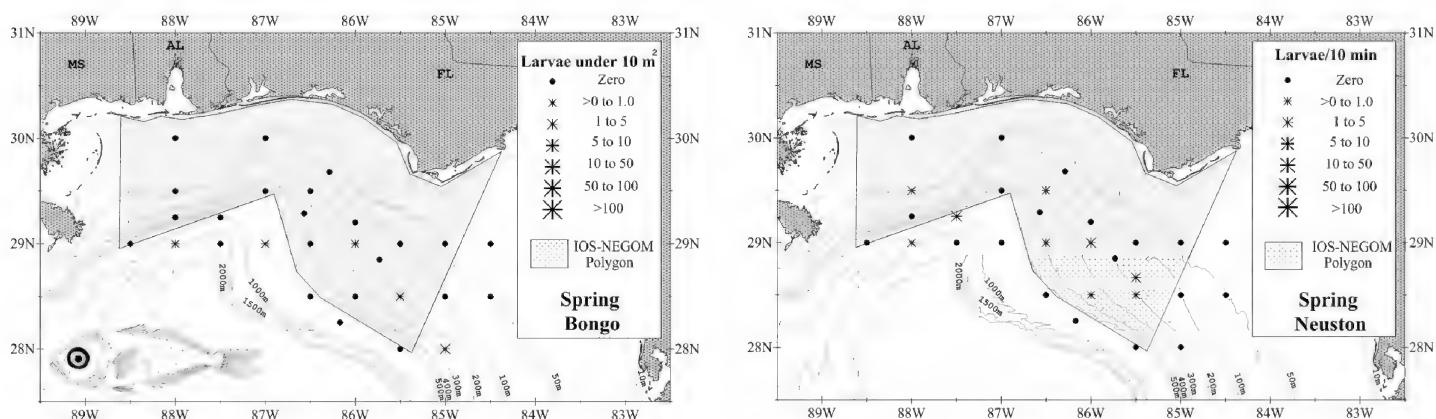

Figure 35. *Mean abundance and mean CPUE of rough scad,* Trachurus lathami, *larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

areas were comparable (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

LUTJANIDAE (190 occurrences; 728 larvae; Figure 36)

Snapper larvae in this taxon were generally < 3.0 mm in length and/or did not exhibit enough dorsal spine development to permit positive identification to genus (Lindeman et. al. 2006; Lyczkowski–Shultz and Hanisko 2007). Larvae were taken predominately in bongo samples and during fall surveys (Table 1). Mean size in bongo samples was 2.4 mm

BL (n = 457; range = 1.3–4.2 mm) and mean size in neuston samples was 3.2 mm BL (n = 32; range = 2.3–5.4 mm). Small, early–stage snapper larvae were ubiquitously distributed with typical mean abundances of 10–50 larvae under 10 m^2 throughout the area (Figure 36). Frequency of occurrence of snapper larvae in the UNIS and Gulfwide survey areas was comparable in both seasons and for both sampling gears (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).
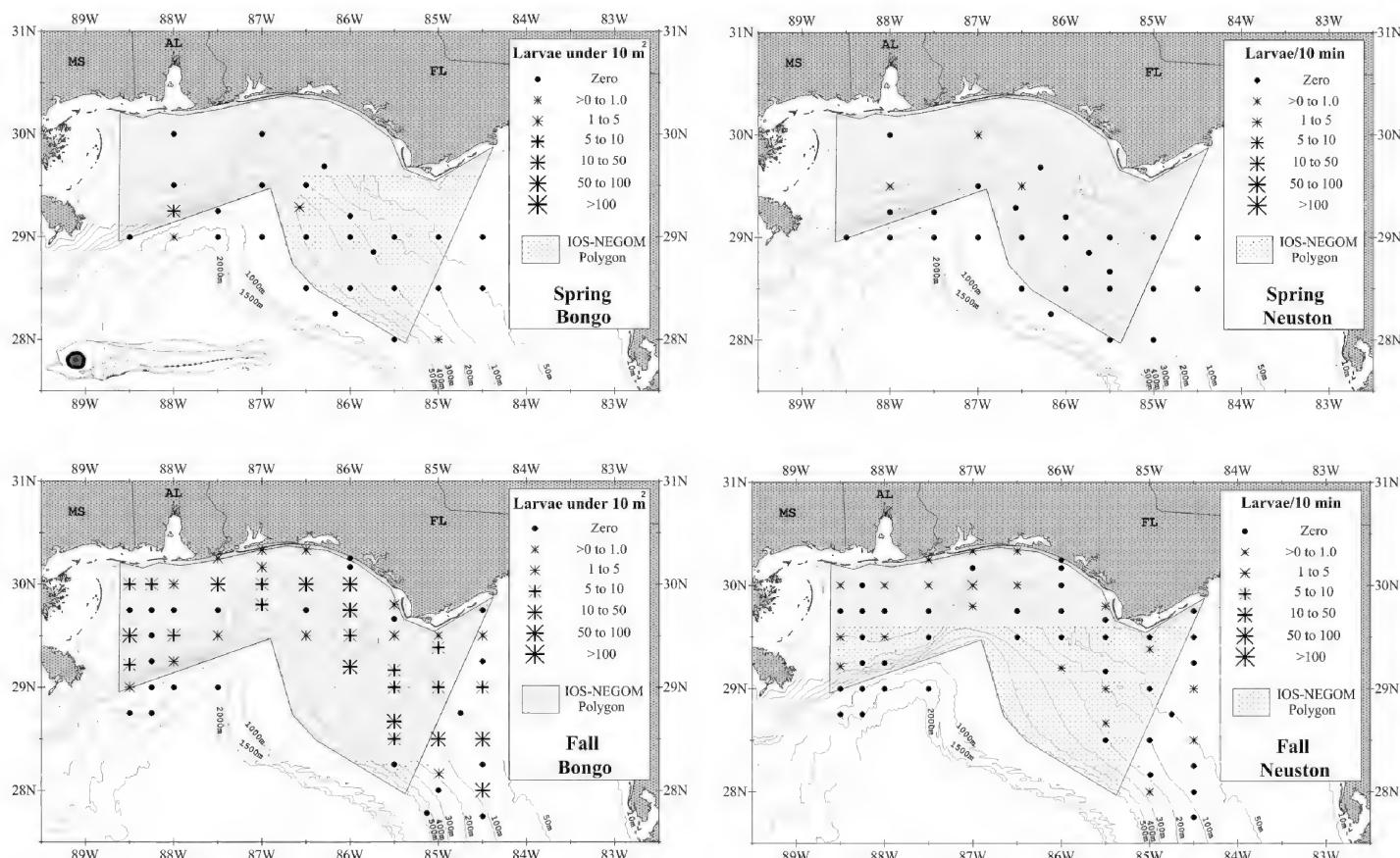
Figure 36. Mean abundance and mean CPUE of snapper (Lutjanidae) larvae (<3.0 mm) at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
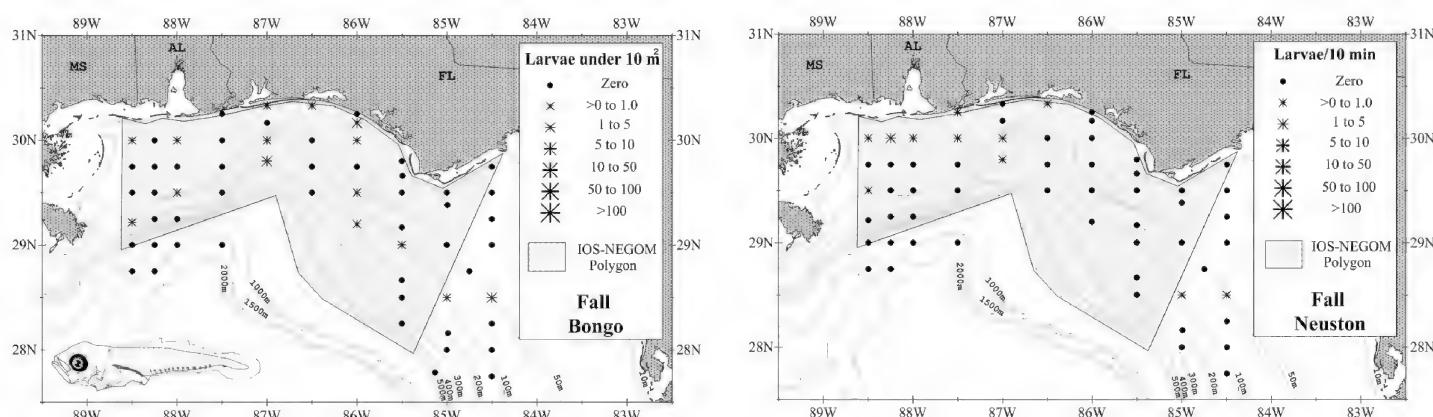
Figure 37. Mean abundance and mean CPUE of snapper larvae, genus Lutjanus, at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.

*Lutjanus* spp. (34 occurrences; 64 larvae; Figure 37)

The larvae in this taxon could be reliably identified only to genus because dorsal spine development was not advanced enough to permit identification to species (Lindeman et. al. 2006; Lyczkowski–Shultz and Hanisko 2007). *Lutjanus* spp. larvae were taken as often in bongo as in neuston samples but were nearly twice as numerous in neuston collections; all specimens were taken during fall surveys (Table 1). Mean size in bongo samples was 3.3 mm BL (n = 20; range = 2.0–7.6 mm) and mean size in neuston samples was 3.5 mm BL (n = 22; range = 2.4–4.9 mm). Most occurrences and specimens were found at stations within the 100 m isobath (Figure 37). *Lutjanus* larvae were significantly less common in UNIS study area bongo and neuston samples than Gulfwide (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

Lutjanus campechanus (33 occurrences; 71 larvae; Figure 38)

Larvae of the red snapper, a commercially important species in the GOM, were taken more often and in greater numbers in neuston than in bongo samples and, except for one occurrence of 2 specimens, were all taken during fall surveys (Table 1). Mean size in bongo samples was 4.6 mm BL (n =

12; range = 3.5–6.3 mm) and mean size in neuston samples was 4.2 mm BL (n = 38; range = 3.2–6.5 mm). Red snapper larvae occurred most often and in greater numbers along or west of 87°W longitude (Figure 38). A single occurrence in spring (not shown in Figure 38) was far offshore at station B001 over water depth of 1500 m. Frequency of occurrence (bongo and neuston samples) and relative abundance (bongo samples) were greater Gulfwide than in the UNIS study area, but relative abundance in neuston samples was comparable (Table 3). In a recent examination of SEAMAP ichthyoplankton data from 1982–2003, Lyczkowski–Shultz and Hanisko (2007) described the seasonal occurrence, distribution, and abundance of red snapper larvae throughout the northern GOM. Red snapper were significantly less common in the UNIS study area than Gulfwide in fall bongo samples (Table 3). Their occurrence in spring bongo and fall neuston samples was comparable in the 2 areas. Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

Lutjanus griseus (9 occurrences; 9 larvae; Figure 39)

Gray snapper larvae were present in both bongo and neuston collections and all occurrences but one were during fall surveys (Table 1). Mean size in bongo samples was 5.0 mm BL (n = 4; range = 4.2–6.6 mm) and mean size in neuston
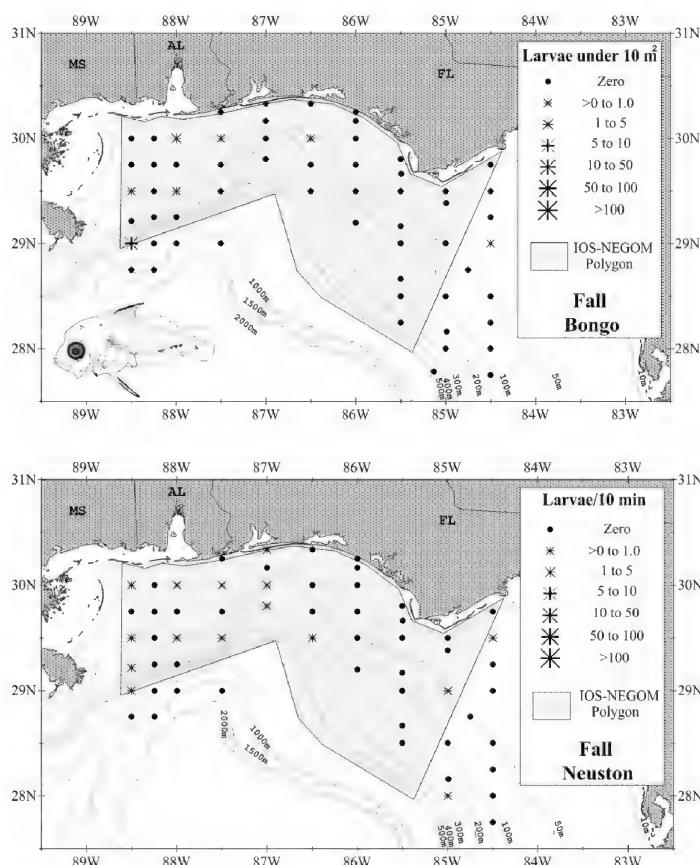
Figure 38. *Mean abundance and mean CPUE of red snapper,* Lutjanus campechanus, *larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. (Map of one occurrence in spring survey neuston samples is not shown.) UNIS and SEAMAP defined in Figure 1.*
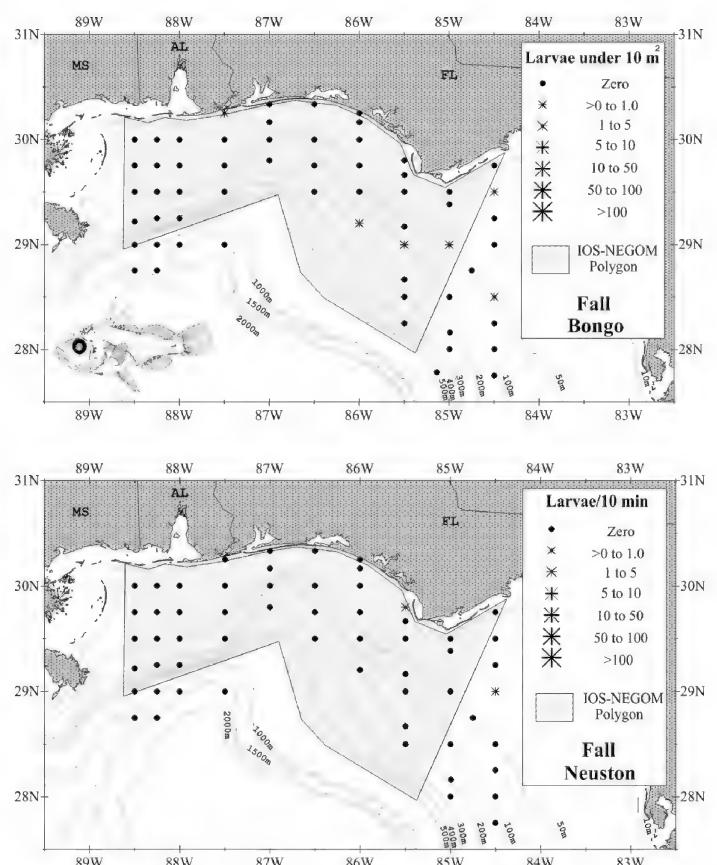

Figure 39. *Mean abundance and mean CPUE of gray snapper,* Lutjanus griseus, *larvae at stations in the UNIS study area captured during SEAMAP surveys, 1982-1999. (Map of one occurrence in spring survey neuston samples is not shown.) UNIS and SEAMAP defined in Figure 1.*

samples was 4.0 mm BL (n = 3; range = 3.5–4.8 mm). Larvae were almost exclusively found at stations in the eastern UNIS study area, along or east of 86°W longitude (Figure 39). The single occurrence in spring (not shown in Figure 39) was at station B172 over 100 m water depth. The occurrence of gray snapper larvae was comparable in the UNIS and Gulfwide survey areas (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

Pristipomoides aquilonaris (74 occurrences; 208 larvae; Figure 40)

Larvae of this small snapper, the wenchmen, were commonly taken in the UNIS study area primarily in bongo samples and almost exclusively during fall surveys (Table 1). Mean size in bongo samples was 3.8 mm BL (n = 113; range = 1.9–9.0 mm) and mean size in neuston samples was 4.3 mm BL (n = 54; range = 2.8–5.7 mm). Wenchmen larvae were distributed in a band along the 50–300 m isobaths outlining the DeSoto Canyon across the full extent of the UNIS study area (Figure 40). The occurrence of wenchmen larvae was comparable in the UNIS and Gulfwide survey areas (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

Rhomboplites aurorubens (174 occurrences; 644 larvae; Figure 41)

Vermilion snapper larvae were the second most frequently taken and abundant among snapper larvae in the UNIS study area; only larvae identified to family were more numerous. Although more total specimens were collected in neuston samples, 67% of all occurrences resulted from bongo net samples (Table 1). Mean size in bongo samples was 4.1 mm BL (n = 254; range = 2.6–11.2 mm) and mean size in neuston samples was 3.8 mm BL (n = 227; range = 2.5–6.8 mm). All but 4 occurrences and 97% of specimens were taken during fall surveys. Vermilion snapper larvae were widely distributed through the study area but were taken more consistently at stations at or east of 87°W longitude (Figure 41). Vermilion snapper larvae were significantly more common in the UNIS than Gulfwide survey area in fall bongo samples, but their occurrence in the 2 areas was comparable during spring surveys and in fall neuston samples (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.2% (Table 4).

LOBOTIDAE

Lobotes surinamensis (23 occurrences; 39 larvae; Figure 42)

Tripletail larvae were taken only in neuston samples and primarily during fall surveys (Table 1). The 2 springtime occurrences were located well off the continental shelf (Figure 42). Mean length of larvae was 9.7 mm (n = 10; range = 6.0–
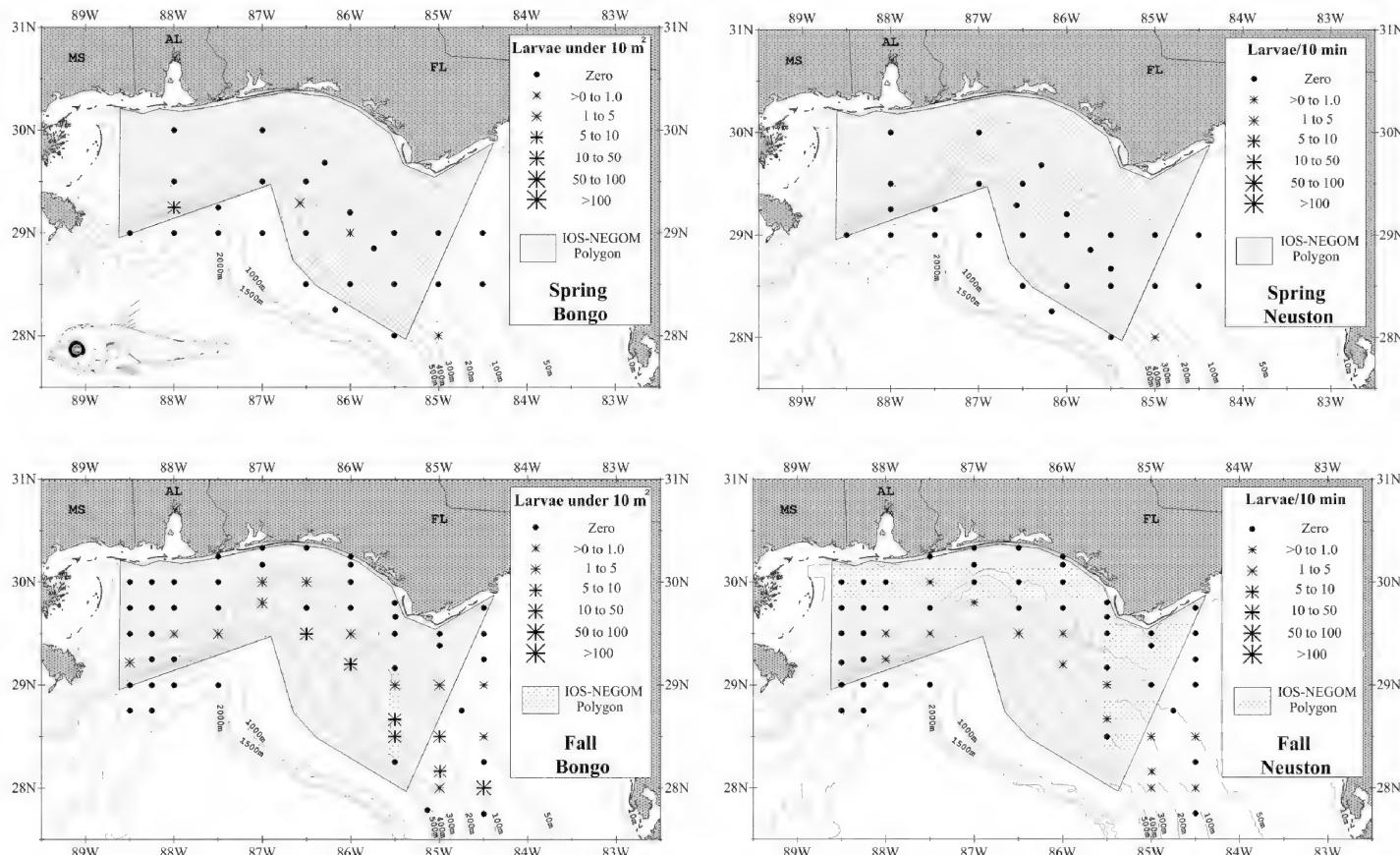
Figure 40. Mean abundance and mean CPUE of wenchman, Pristipomoides aquilonaris, larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
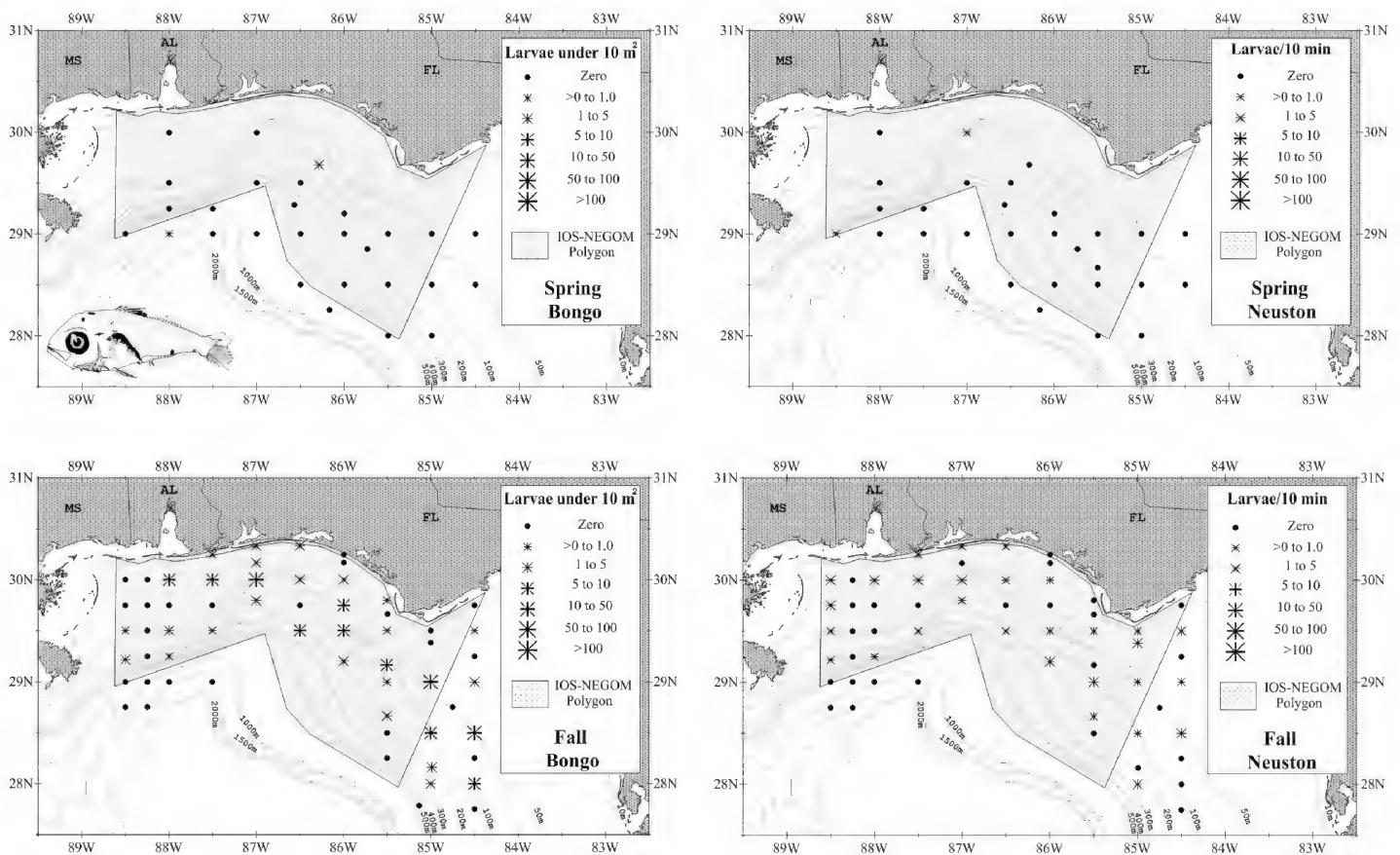
Figure 41. *Mean abundance and mean CPUE of vermilion snapper,* Rhomboplites aurorubens, *larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*
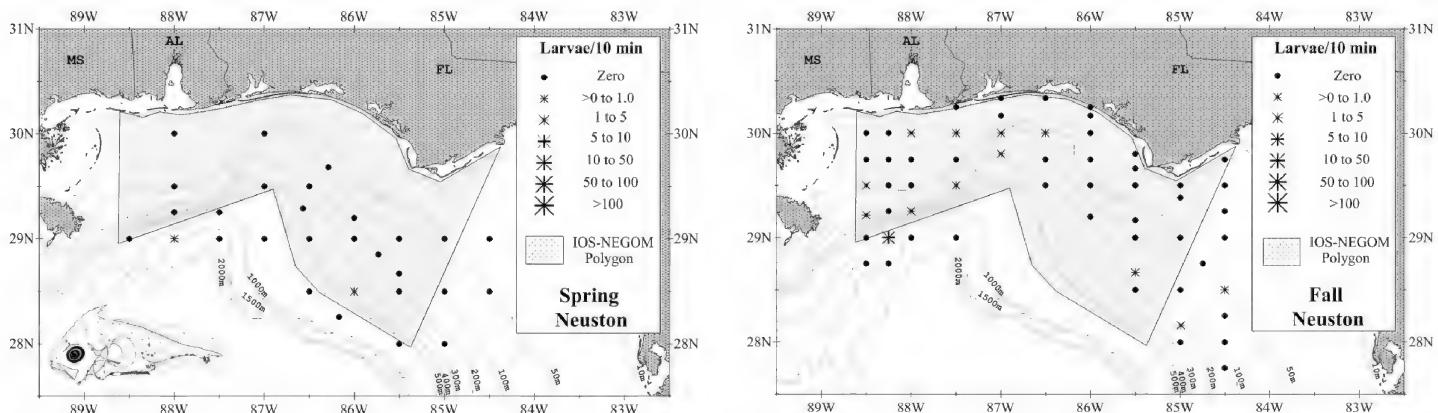
Figure 42. *Mean CPUE of tripletail,* Lobotes surinamensis, *larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

17). Tripletail larvae were more consistently taken and more abundant in the western region of the UNIS study area, along or west of 87°W longitude. Frequency of occurrence of tripletail larvae was comparable in the UNIS and Gulfwide survey areas (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

HAEMULIDAE (10 occurrences; 139 larvae; Figure 43)

Grunts are important predators on offshore reefs through-out the GOM (Hoese and Moore 1977). Larvae hatch from pelagic eggs in a relatively undeveloped state and early stage larvae are difficult to distinguish from the larvae of many other percoid families (Leis and Rennis 1983; Lindeman and Richards 2006). There is no specialized pelagic juvenile in the early life history of grunts and it appears that grunt larvae are not widely dispersed but settle to bottom habitats within 13–20 d of hatching at 6.5–9 mm in length (Linde-
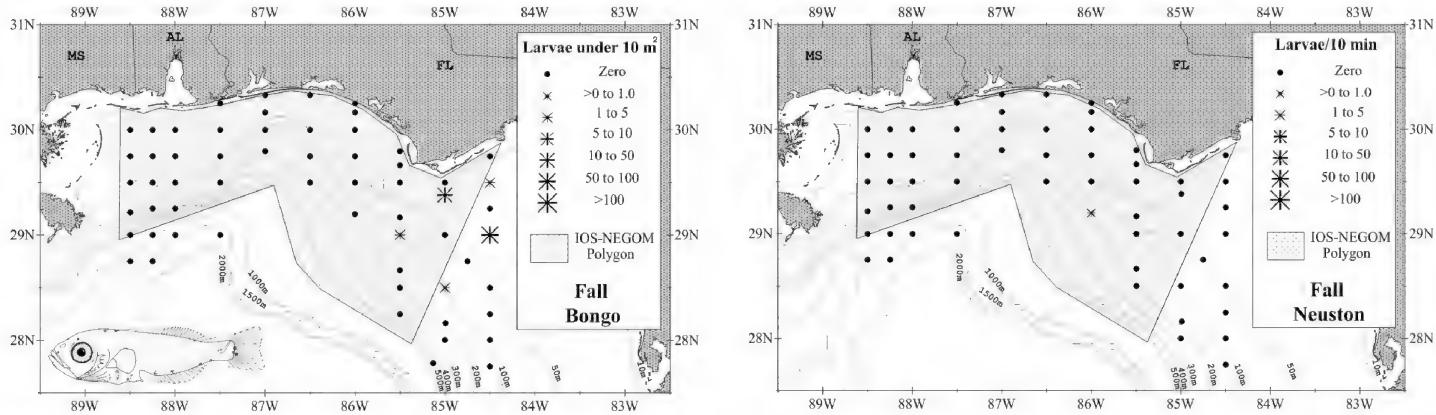
Figure 43. *Mean abundance and mean CPUE of grunt (Haemulidae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. (Map of 2 occurrences in spring survey neuston samples is not shown.) UNIS and SEAMAP defined in Figure 1.*
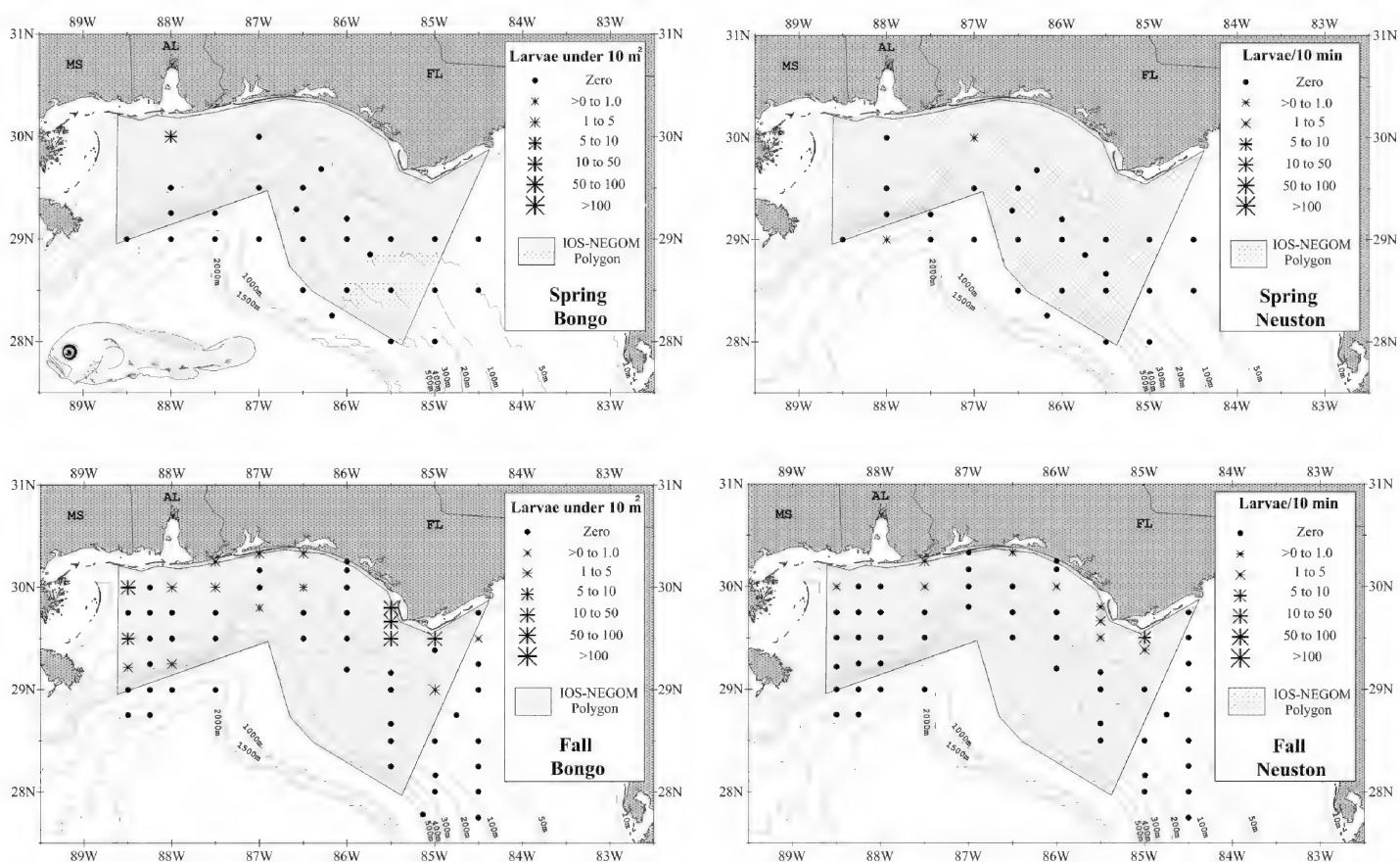
**Figure 44.** *Mean abundance and mean CPUE of seatrout (Cynoscion spp.) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

man et al. 2001). Grunt larvae were taken mostly in bongo samples and almost exclusively during fall surveys (Table 1). Mean size in bongo samples was 3.5 mm BL (n = 6; range = 2.0–6.0 mm) and mean size in neuston collections was 3.3 mm BL (n = 3; range = 2.8–3.7 mm). Haemulid larvae occurred 3 times in neuston collections between longitudes 87 and 86°W in the vicinity of the head of the DeSoto Canyon over 100–200 m water depth. However, most occurrences and highest mean abundances were at stations east of 86°W longitude and inshore of the 100 m isobath (Figure 43). This

pattern corresponds with the general and striking absence of grunts from the list of species inhabiting the Pinnacles deep reefs off Alabama and Mississippi (Weaver et. al. 2002). There were 2 occurrences (not shown in Figure 43) in spring neuston samples at stations B172 and U015 over 100 m water depth. Frequency of occurrence of grunt larvae was comparable in the UNIS and Gulfwide survey areas (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

SCIAENIDAE

80

Cynoscion spp. (64 occurrences; 515 larvae; Figure 44)

This taxon is comprised of the larvae of *Cynoscion arenarius* (sand seatrout) and *C. nothus* (silver seatrout). Although spawning is somewhat separated in time and space, the larvae of these 2 species of sciaenids are difficult to distinguish from each other. Over 80% of sea trout larvae were taken in bongo samples and most during fall surveys (Table 1). Mean size in bongo samples was 2.7 mm BL (n = 160; range = 1.2–6.6 mm) and mean size in neuston collections was 3.9 mm BL (n = 52; range = 1.8–6.1 mm). Sea trout larvae consistently occurred inshore of the 200 m isobath with the highest mean abundances being found inshore in the northeastern corner of the IOS–NEGOM research polygon, i.e. around Cape San Blas (Figure 44). *Cynoscion* spp. larvae occurred significantly less frequently in the UNIS study area than Gulfwide during fall surveys (Table 3). Larvae were taken significantly more frequently in spring neuston samples in the UNIS study area than Gulfwide but at comparable frequency in spring bongo samples. Relative abundances and CPUEs in the 2 areas differed by < 2.5% (Table 4).

Sciaenops ocellata (48 occurrences; 351 larvae; Figure 45)

The larvae of the late summer to fall spawning red drum were taken in both bongo and neuston samples during fall surveys only (Table 1). Mean size in bongo samples was 2.4 mm BL (n = 110; range = 1.4–5.2 mm) and mean size in neuston collections was 3.8 mm BL (n = 46; range = 2.7–6.7 mm). Along or west of longitude 87°W, red drum larvae occurred inshore of the 200 m isobath while east of that meridian larvae were only found inshore of the 50 m isobath (Figure 45). Red drum larvae occurred significantly less frequently in the study area than Gulfwide (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

MULLIDAE (268 occurrences; 19,855 larvae; Figure 46)

The young of the bottom–dwelling, reef associated goatfishes are abundant in the surface waters of the NEGOM and were among the most numerous of taxa collected in SEAMAP neuston collections. Goatfishes pass through a pelagic juvenile stage when they superficially resemble young mullet and occupy the same niche in offshore surface waters. Goatfish young were almost exclusively taken during spring surveys in the UNIS study area (Table 1). Larvae ranged in size from 2.5–18 mm in bongo samples and 2.4–44.5 in neuston samples. Larvae were distributed throughout the area during spring with the highest mean abundances of 100–500 larvae/10 min at 2 stations on the 200 m isobath between longitudes 86.5 and 86°W (Figure 46). Larvae were significantly more common in the UNIS study area than Gulfwide during spring surveys but occurrence during fall surveys was comparable in the 2 areas (Table 3). Relative abundance and CPUE in the 2 areas differed by < 0.2% in spring bongo samples and during fall surveys, whereas, the relative CPUE in UNIS study area fall neuston samples exceeded the Gulfwide value by 20.4% (Table 4). This was disproportionately higher than expected given the fewer samples taken in the study area.

CHAETODONTIDAE (11 occurrences; 12 larvae; Figure 47)

Butterflyfishes are among the most characteristic members of reef fish communities. Their larvae are distinctive especially after formation of the unique bony head plates that mark their specialized, pelagic 'tholichthys' stage (Leis and Rennis 1983, Kelley 2006). Chaetodontid larvae are a rare component of plankton collections and tend to be more numerous in distant oceanic waters than near the adult reef habitat (Leis 1989). In the UNIS study area chaetodontid larvae were slightly more common in neuston than bongo samples and over 80% of occurrences and specimens were captured during fall surveys (Table 1). Mean size in neuston collections was 4.8 mm BL (n = 5; range = 3.0–8.2 mm) and mean size in bongo samples was 3.6 mm BL (n = 5; range = 2.5–5.2 mm). Larvae were taken only at stations along or east of longitude 86.5°W but were found over water depths ranging from < 50 to > 500 m (Figure 47). Butterflyfish larvae were significantly less common in spring neuston samples in the study area than Gulfwide (Table 3). Their occurrence in
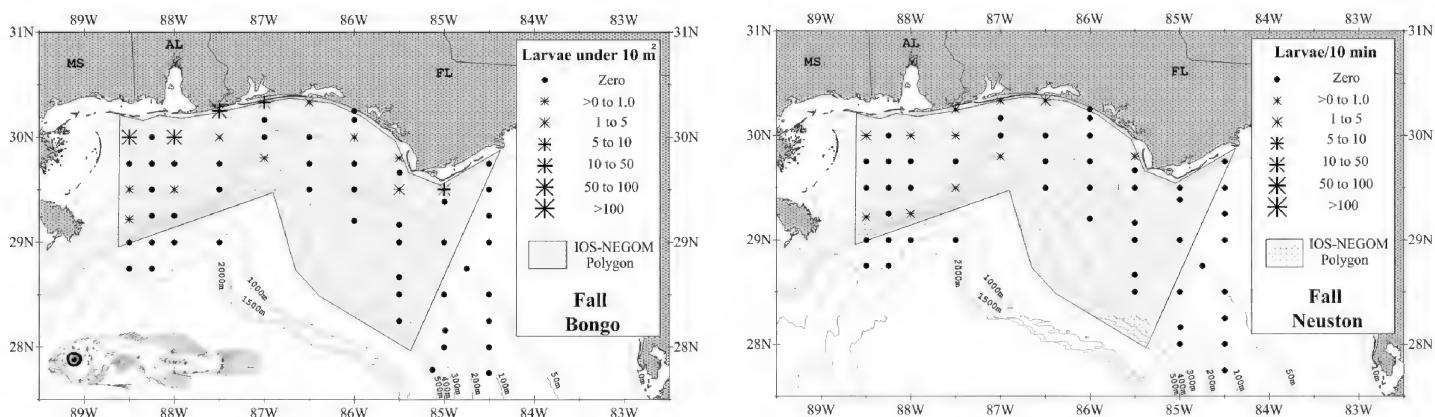
Figure 45. *Mean abundance and mean CPUE of red drum, Sciaenops ocellata, larvae at stations in the UNIS study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*
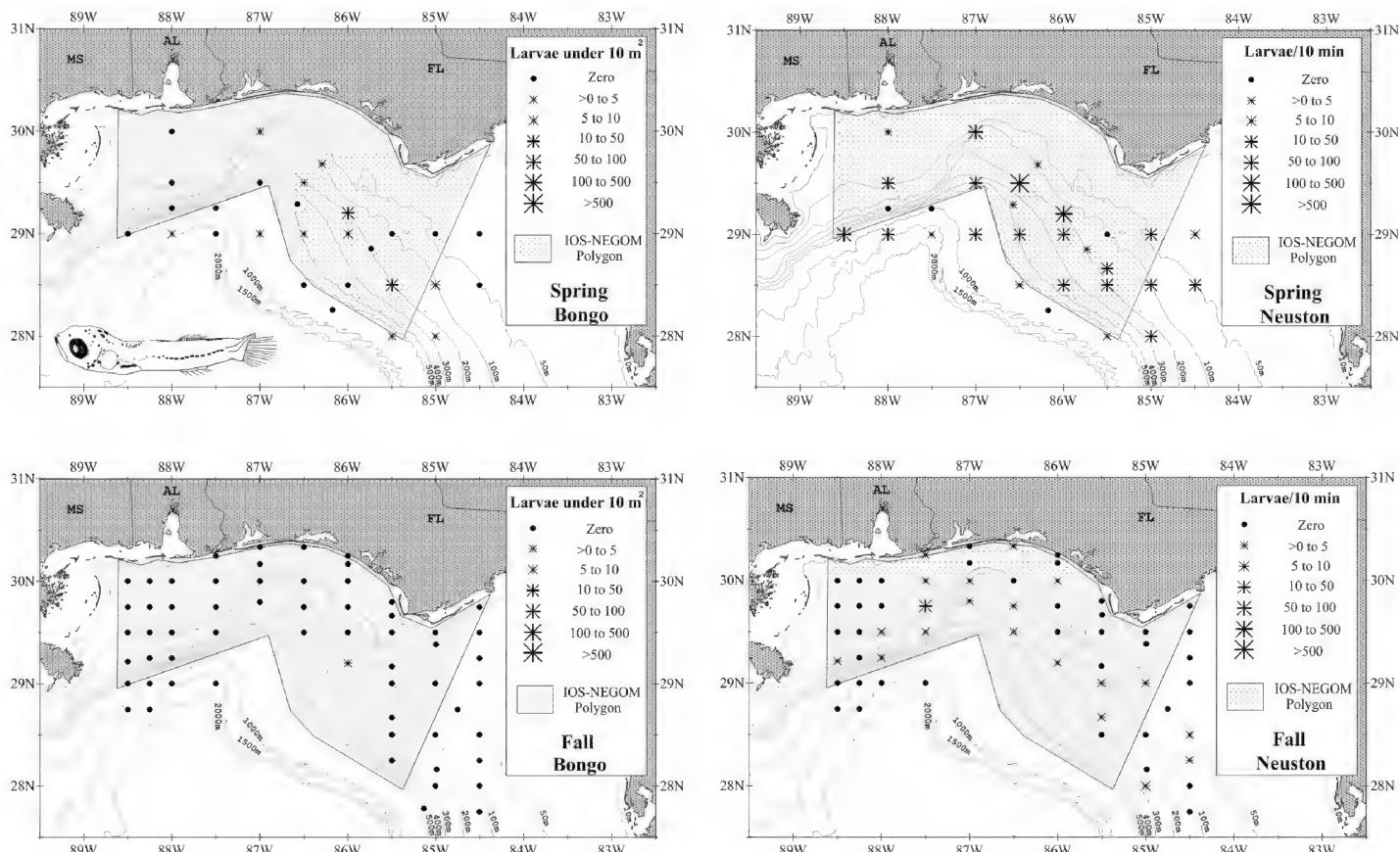
**Figure 46.** Mean abundance and mean CPUE of goatfish (Mullidae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
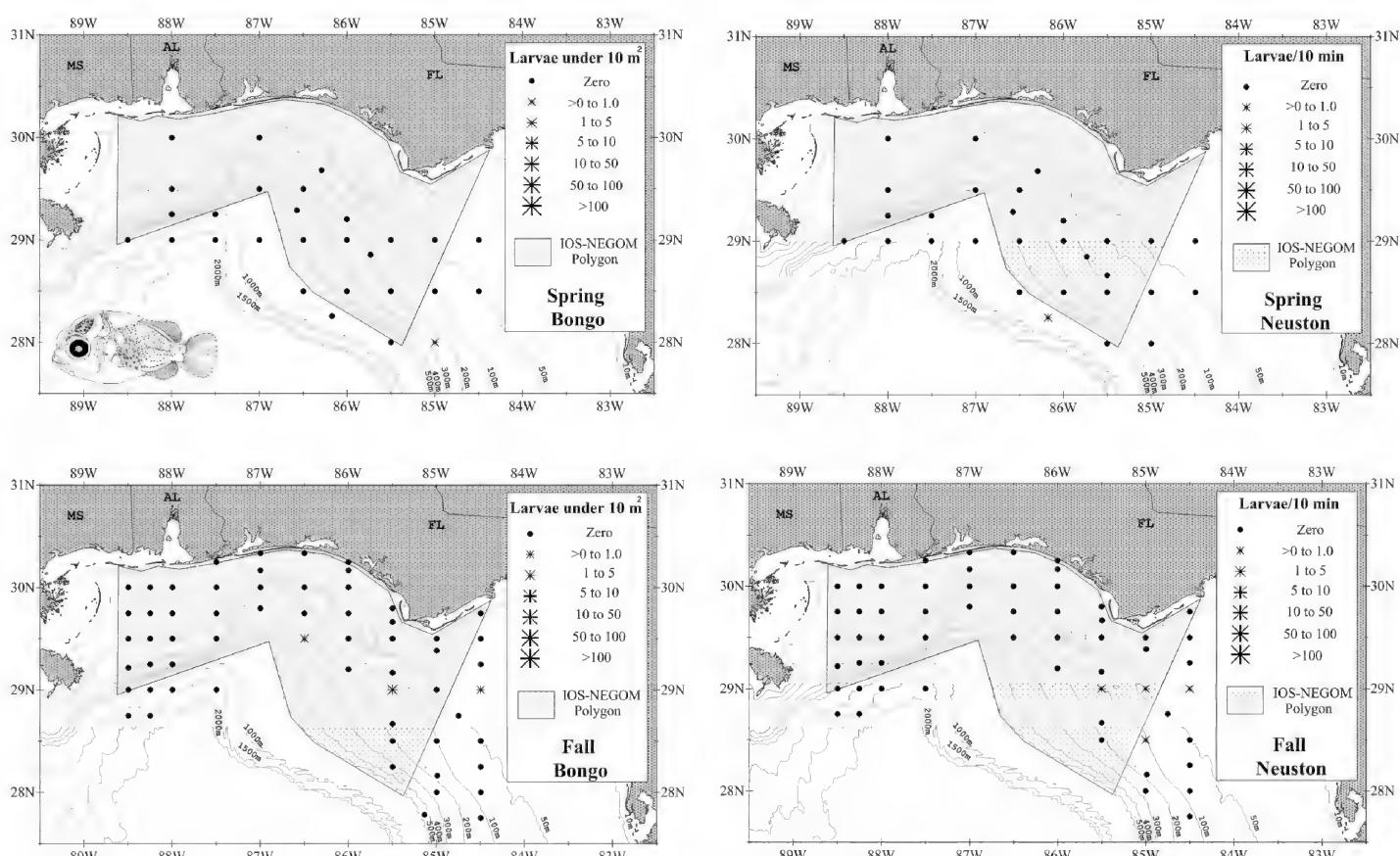

Figure 47. Mean abundance and mean CPUE of butterflyfish (Chaetodontidae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.

spring and fall bongo samples was comparable in both areas. However, despite the few captures overall, occurrence in fall neuston samples was significantly higher in the study than Gulfwide. Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

POMACANTHIDAE (6 occurrences; 7 larvae; Figure 48)

Angelfishes are also characteristic members of reef communities. Their larvae, like those of the previous family, are distinctive and rare in plankton collections but the angelfishes do not have a specialized pelagic stage. Pomacanthid larvae were taken in both neuston and bongo samples, and during spring and fall surveys (Table 1). There were too few occurrences overall to suggest spatial or seasonal differentiation within the UNIS study area. Mean size in neuston collections was 10.1 mm BL (n = 4; range = 3.1–14.0 mm) and mean size in bongo samples was 6.1 mm BL (n = 3; range = 3.8–7.2 mm). Distribution of angelfish (Figure 48) was similar to that of butterflyfish larvae (Figure 47). Angelfish larvae were taken only at stations along or east of longitude 86.5°W and were found over water depths ranging from 50–400 m. The frequency of occurrence of larvae was not significantly different between the study area and Gulfwide (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

POMACENTRIDAE (63 occurrences; 166 larvae; Figure 49)

The damselfishes are among the most studied reef fishes, yet the larval development of few species has been described. The eggs of these fishes are demersal but the larvae of most species are planktonic (Leis and Rennis 1983, Watson 1996b). Identification even to the family level remains problematic for the Pomacentridae (Kavanagh et al. 2000). This is especially true in the GOM where the poorly known larvae of mullids, gerreids and sparids are abundant. Larvae of these perciform families closely resemble pomacentrid larvae. For example, juvenile *Abudefduf saxatilis* were found during a recent re—examination of specimens identified as Sparidae (porgies) in SEAMAP collections. Despite uncertainties, SEAMAP data on larval pomacentids are summarized here under the caveat that misidentifications have led to an underestimation of occurrence and abundance of at least one pomacentrid (as noted above). Damselfish larvae were taken as often in neuston as in bongo nets and, although the total number of specimens was equally divided between spring and fall surveys, larvae occurred 3 times more frequently in fall survey samples than in spring survey samples (Table 1). Mean size in neuston collections was 4.9 mm BL (n = 30; range = 2.5–17.8 mm) and mean size in bongo samples was 2.9 mm BL (n = 37; range = 1.6–5.0 mm). Larvae were distributed throughout the UNIS study area especially during fall surveys when highest mean abundances were found
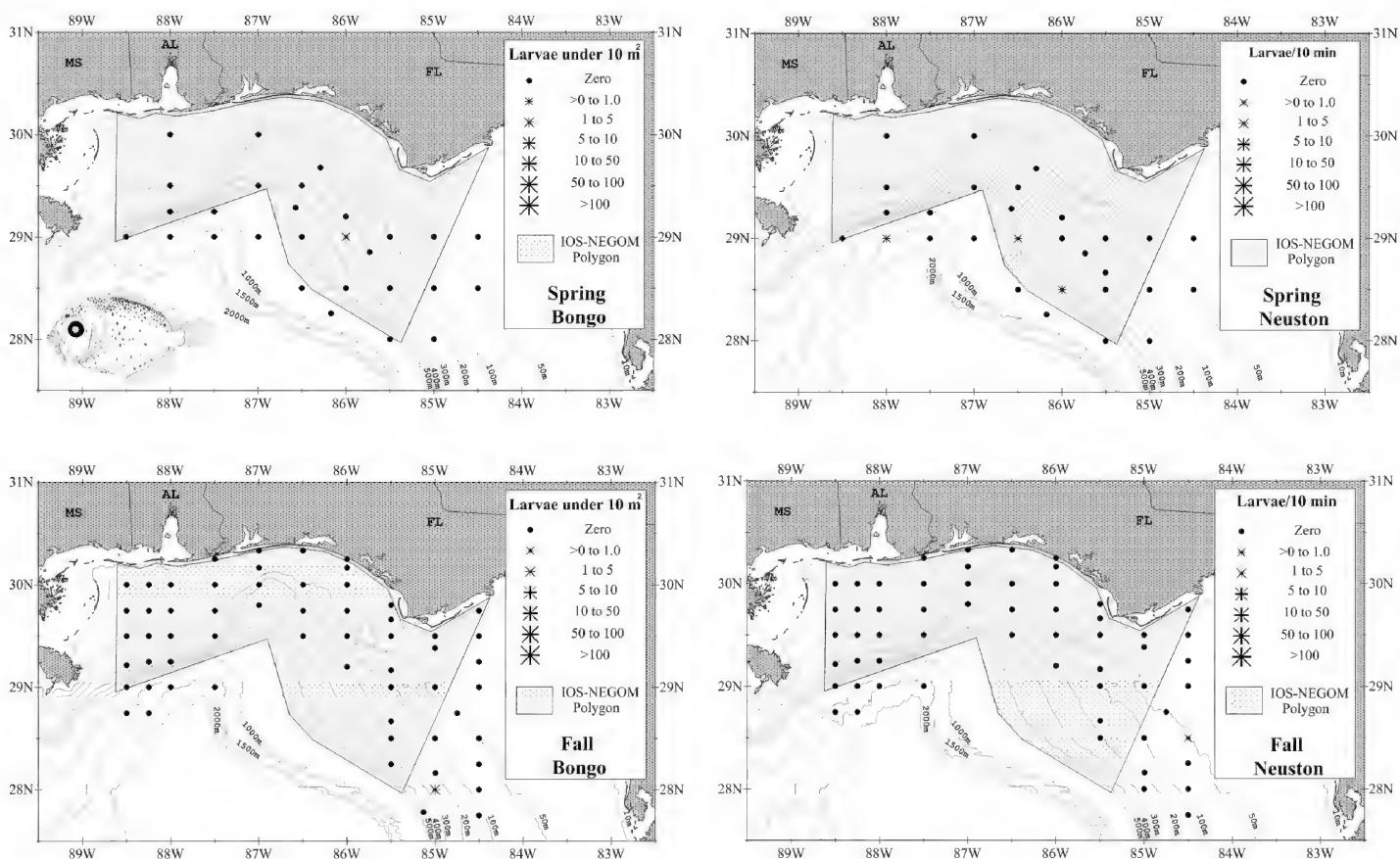
Figure 48. Mean abundance and mean CPUE of angelfish (Pomacanthidae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
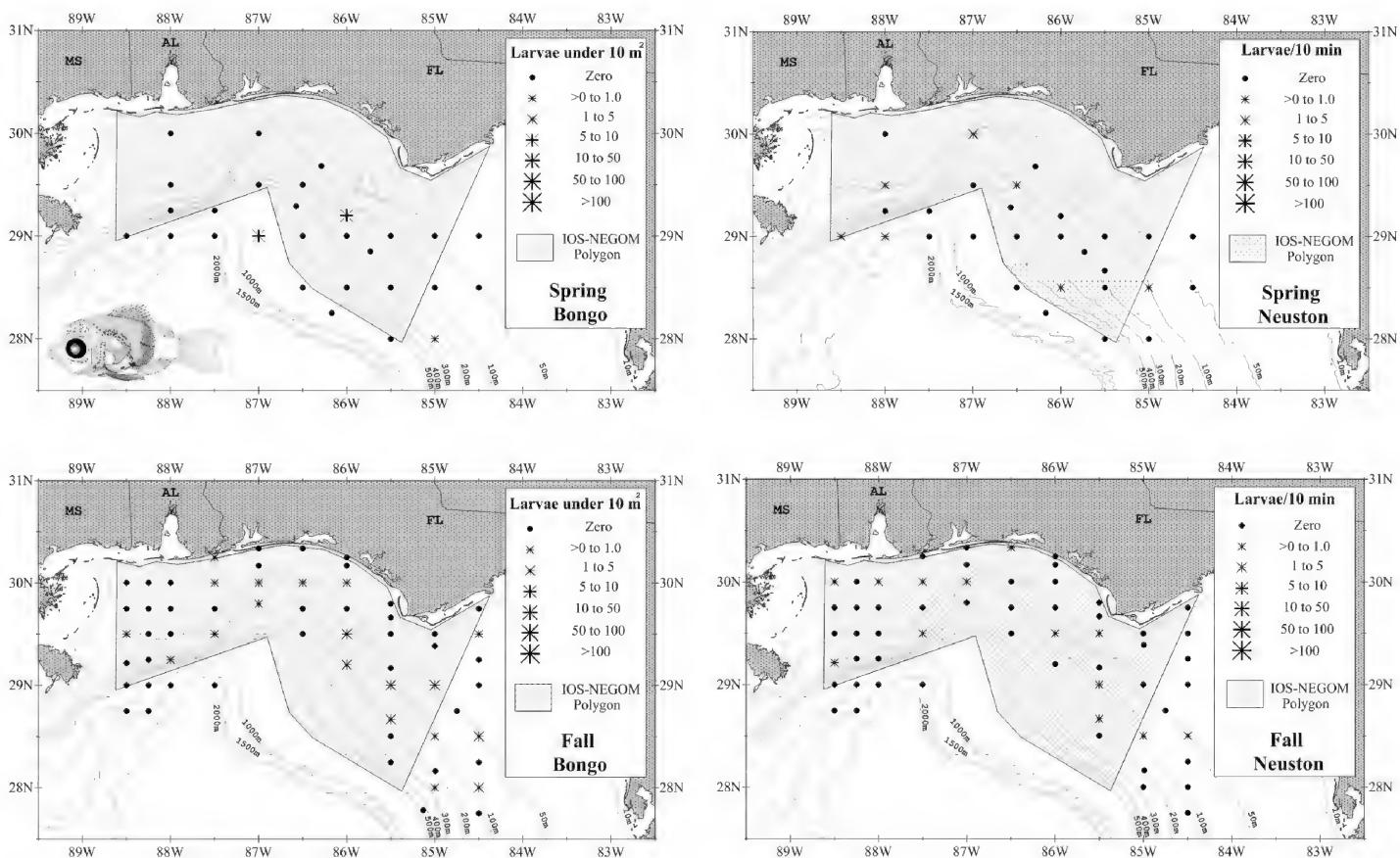
Figure 49. *Mean abundance and mean CPUE of damselfish (Pomacentridae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

along or east of longitude 86.5°W (Figure 49). Damselfish larvae occurred significantly less frequently in spring bongo and fall neuston samples in the study area than Gulfwide but as frequently as Gulfwide in spring neuston and fall bongo samples (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

LABRIDAE (358 occurrences; 3,420 larvae; Figure 50)

Among specimens that could be reliably identified at least to family, larvae of the wrasses were the most numerous among obligate reef fishes in UNIS study area collections. Over 80% of occurrences and 93% of specimens were captured in bongo samples and 86% of occurrences and 95% of specimens were taken during fall surveys (Table 1). Mean size in neuston collections was 7.1 mm BL (n = 54; range = 2.0–12.6 mm) and mean size in bongo samples was 5.0 mm BL (n = 140; range = 1.2–12.0 mm). Wrasse larvae were homogeneously distributed throughout the study area, being taken at nearly every station during fall surveys (aside from one stations within the IOS–NEGOM research polygon and 4 stations in the extreme southwestern corner of the UNIS study area; Figure 50). Frequency of occurrence of wrasse larvae in study area samples (72%) far exceeded their occurrence Gulfwide (41%) during fall surveys when larvae were statistically far more common in the study area than Gulfwide in both bongo and neuston samples (Table 3). Frequency of occurrence in the study area was either significantly less (bongo

samples) than Gulfwide or comparable (neuston samples) to Gulfwide occurrence during spring surveys. Relative abundance and CPUE in the 2 areas differed by < 0.2% except in fall bongo samples, when the relative abundance was 2.5% higher than expected in the study area than Gulfwide (Table 4).

SCARIDAE (113 occurrences; 369 larvae; Figure 51)

Although not as abundant as wrasse larvae, parrotfish larvae were also taken primarily in bongo net samples during fall surveys (Table 1). Mean size in neuston collections was 7.8 mm BL (n = 27; range = 2.1–11.7 mm) and mean size in bongo samples was 5.4 mm BL (n = 72; range = 1.8–11.0 mm). Parrotfish larvae were not as widely distributed throughout the UNIS study area as labrid larvae and were more frequently taken and more numerous at stations east of longitude 87°W during fall surveys (Figure 51). There was no statistical difference between the UNIS and Gulfwide survey areas in frequency of occurrence of scarid larvae in spring neuston samples or fall survey samples (Table 3). Scarid larvae were taken significantly less frequently in the study area than Gulfwide in spring bongo samples. Relative abundance and CPUE in the 2 areas differed by < 0.1% (Table 4).

ACANTHURIDAE (4 occurrences; 5 larvae; Figure 52)

As in a number of other reef fish families, the duration of the pelagic phase of the surgeonfishes may be long and includes a specialized, pre–settlement stage called the 'ac-
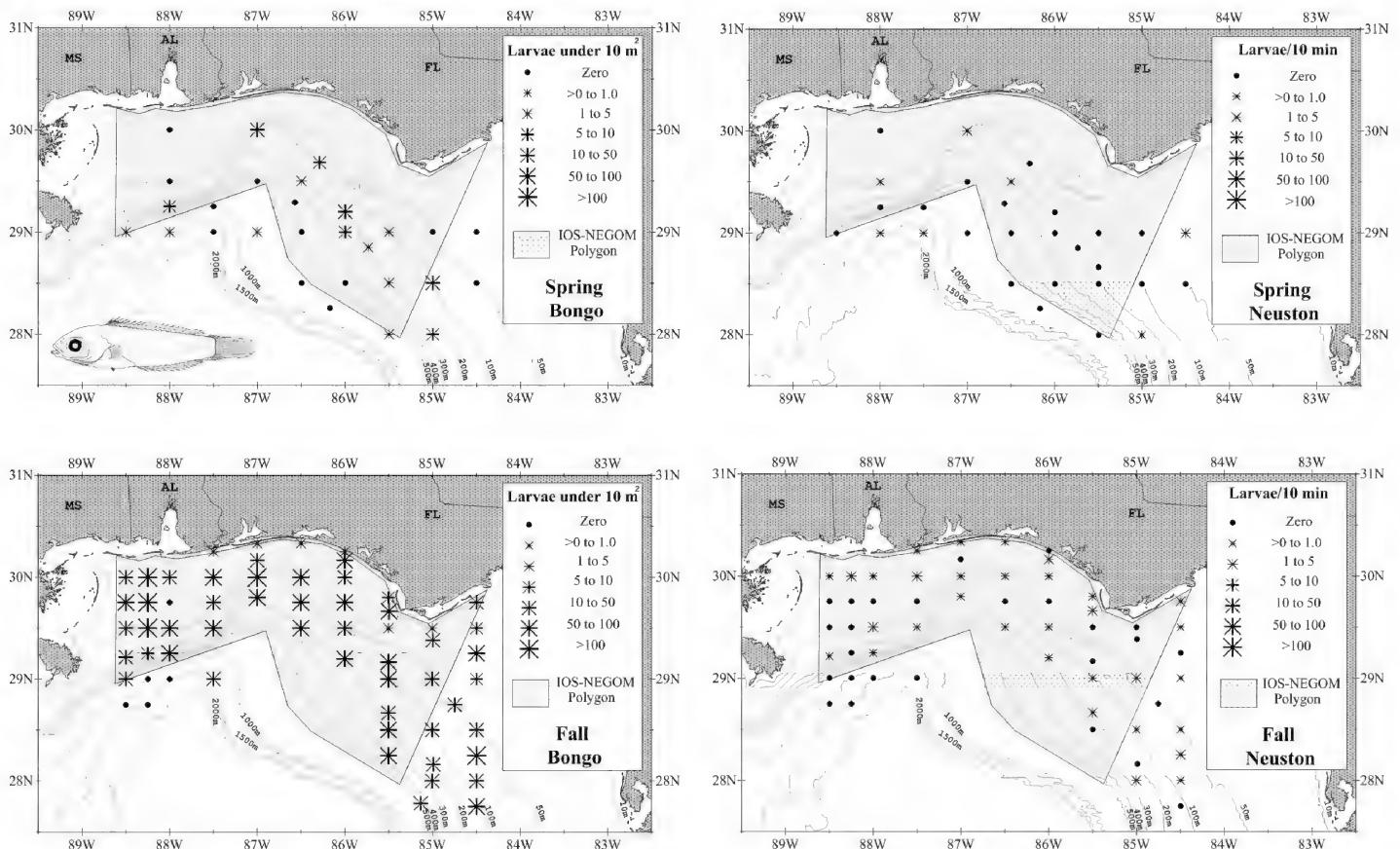
Figure 50. Mean abundance and mean CPUE of wrasse (Labridae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
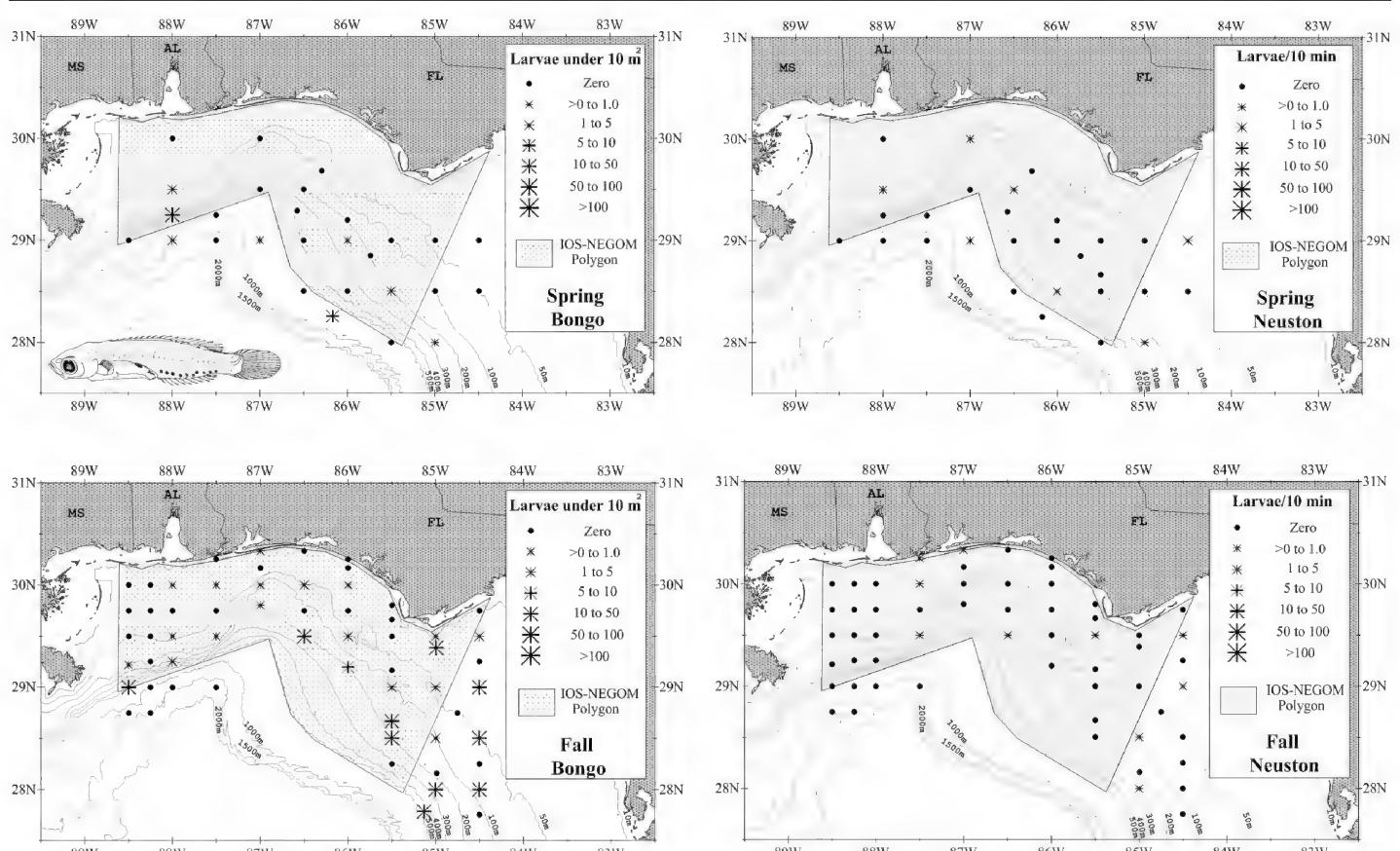
**Figure 51.** Mean abundance and mean CPUE of parrotfish (Scaridae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
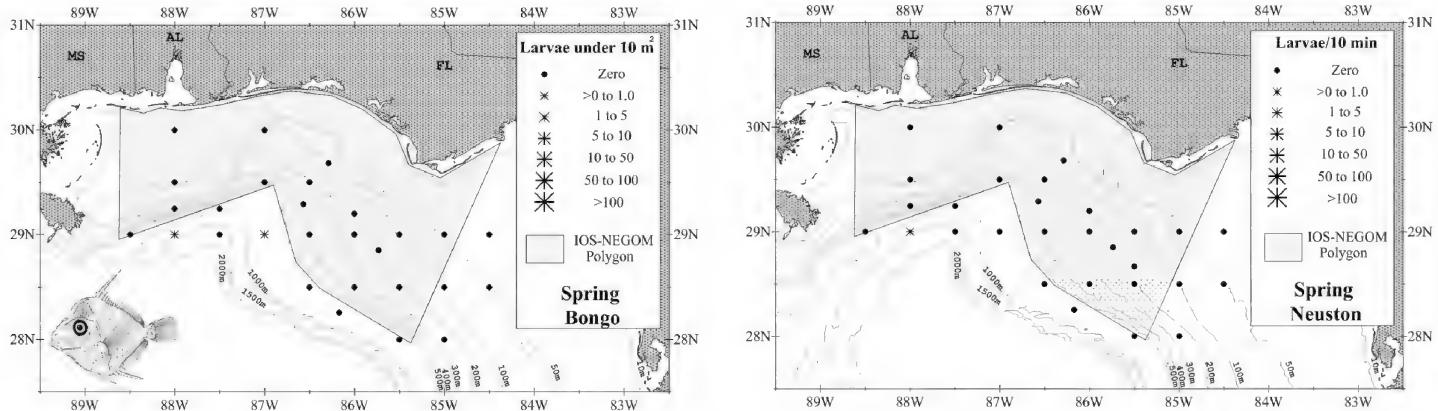
Figure 52. Mean abundance and mean CPUE of surgeonfish, (Acanthuridae) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. (Map of one occurrence in fall survey neuston samples is not shown.) UNIS and SEAMAP defined in Figure 1.
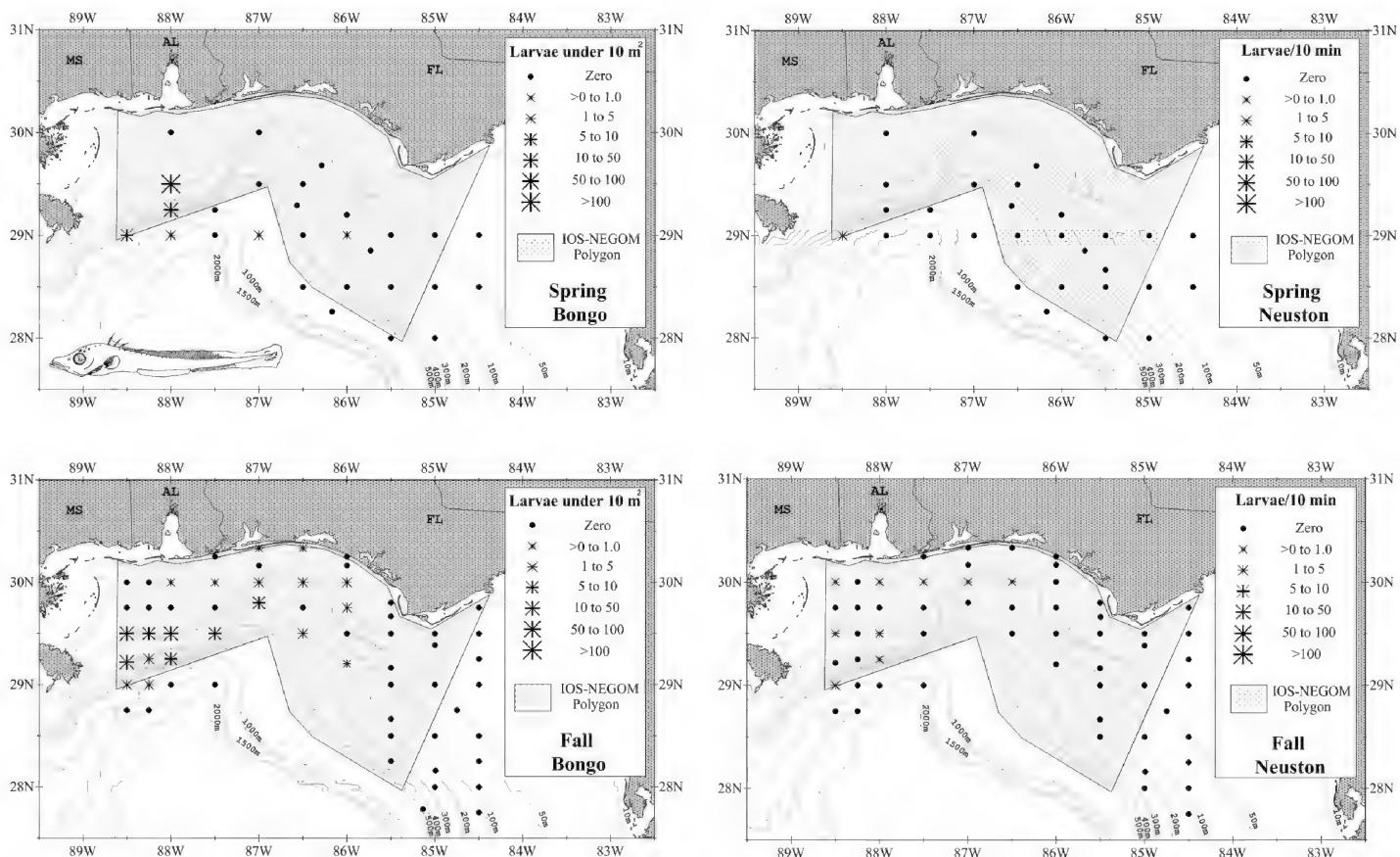
Figure 53. Mean abundance and mean CPUE of Atlantic cutlassfish, Trichiurus lepturus, larvae at stations in the UNIS study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.

ronurus' stage (Thresher 1984). These larvae were rare in the UNIS study area, occurring in both bongo and neuston samples with all but one occurrence during spring surveys (Table 1). The single fall occurrence (not shown in Figure 52) was in a neuston net sample taken at station B153 between the 200 and 300 m contours southwest of the Florida Middle Grounds. Mean size in neuston collections was 3.6 mm BL (n = 3; range = 3.5–3.7 mm). The 2 specimens in bongo samples measured 4.0 and 14.3 mm BL. All captures were made outside the IOS–NEGOM research polygon, at or beyond the continental slope (Figure 52). Frequency of occurrence in the study area was significantly less than Gulfwide during spring surveys, but during fall surveys surgeonfish larvae were as common in neuston samples in the study area as they were in Gulfwide samples (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

TRICHIURIDAE

Trichiurus lepturus (82 occurrences; 260 larvae; Figure 53)

The Atlantic cutlassfish is the most common member of this family in the GOM. Young *T. lepturus* were most frequently captured in bongo samples during fall surveys (Table 1). Mean size in bongo samples was 6.4 mm BL (n = 125; range

= 2.4–26 mm) and mean size in neuston collections was 6.3 mm BL (n = 27; range = 4.3–9.3 mm). Larvae occurred more often and in greater numbers in the central and western regions of the UNIS study area (Figure 53). Atlantic cutlassfish young were never taken east of longitude 86°W. Cutlassfish larvae were significantly more common in spring bongo samples in the UNIS study area than Gulfwide but were as common in both areas in spring neuston samples and during fall surveys (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

SCOMBRIDAE

Acanthocybium solandri (2 occurrences; 2 larvae; Figure 54)

The larvae of another highly prized sport fish, the wahoo, were rare in the UNIS study area and were taken exclusively in bongo net samples during fall surveys (Table 1). These 2 specimens measured 2.8 and 5.0 mm in length. Both stations where larvae were taken lie outside the IOS–NEGOM research polygon (Figure 54). The westernmost capture station was located over water depths >1500 m, whereas the easternmost capture station was located between the 50–100 m isobaths. There was no statistical difference between the UNIS and Gulfwide survey areas in frequency of occurrence of wahoo larvae (Table 3). Relative abundances in the 2 areas differed by <0.1% (Table 4).

Figure 54. Mean abundance of wahoo, Acanthocybium solandri, *larvae (left); and mean CPUE of swordfish*, Xiphias gladius (right), *larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*
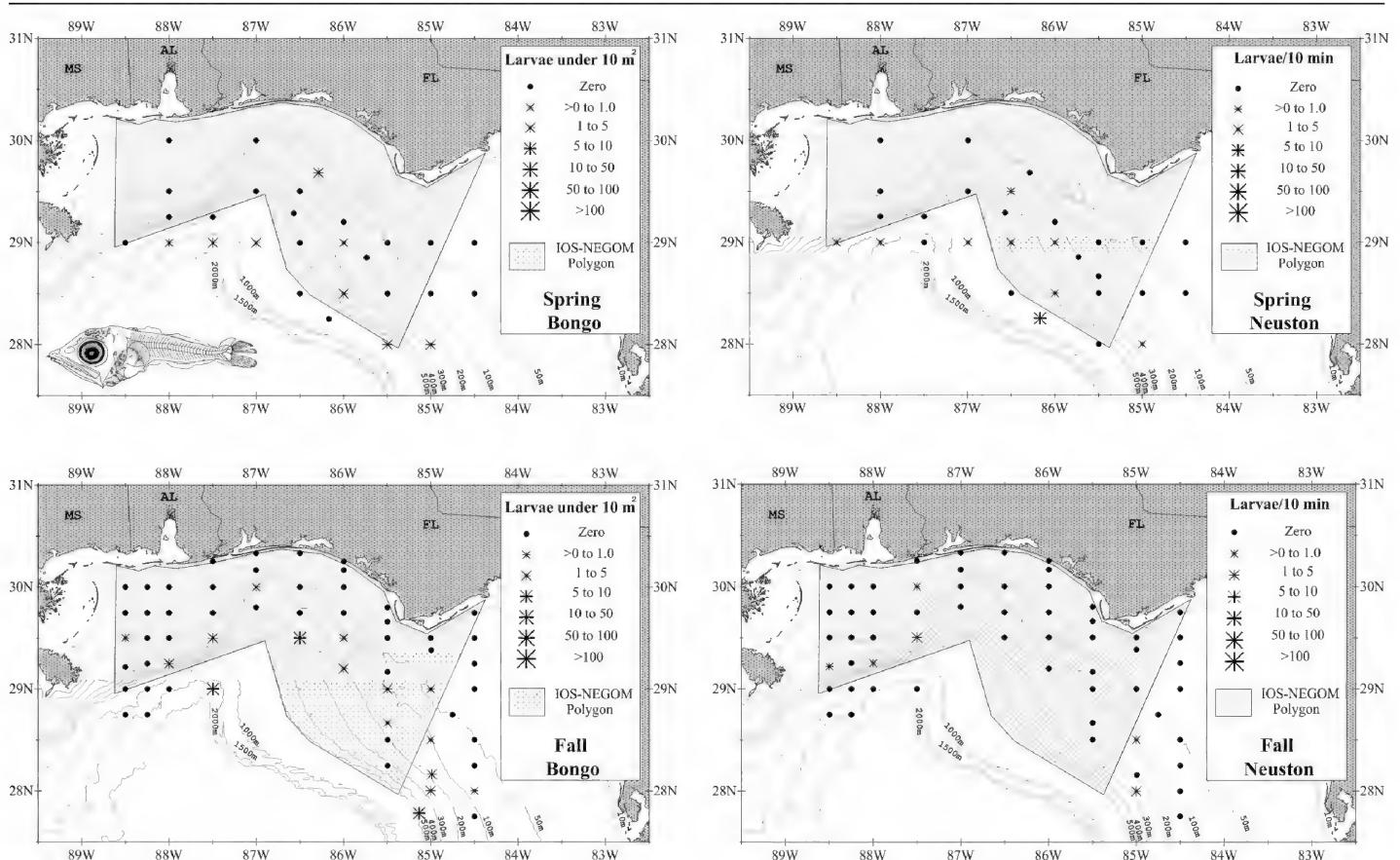
Figure 55. Mean abundance and mean CPUE of skipjack tuna, Katsuwonus pelamis, *larvae at stations in the UNIS study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*

Katsuwonus pelamis (63 occurrences; 136 larvae; Figure 55)

Larvae of skipjack tuna, an oceanic schooling scombrid, were as common in bongo as in neuston samples during spring and fall surveys (Table 1). Mean size in bongo samples was 3.9 mm BL (n = 29; range = 2.3–7.0 mm) and mean size in neuston collections was 5.1 mm BL (n = 46; range = 3.1–9.8 mm). Skipjack tuna larvae were taken most frequently and at the highest mean abundances at stations beyond the 100 m isobath (Figure 55). Skipjack tuna larvae were significantly less common in the study area during spring surveys than Gulfwide (Table 3). During fall surveys there was no statistical difference between the UNIS and Gulfwide survey areas in their occurrence. Relative abundances in the 2 areas differed by < 0.1% (Table 4).

Scomberomorus cavalla (87 occurrences; 143 larvae; Figure 56)

King mackerel larvae occurred across the UNIS study area inshore of the 200 m isobath during fall surveys, but were taken in other regions of the GOM during spring surveys (Table 3). Larvae were captured as often in bongo as in neuston samples (Figure 56); however, neuston collections accounted for 62% of specimens captured (Table 1). Mean size in bongo samples was 3.8 mm BL (n = 50; range = 1.9–7.2 mm) and mean size in neuston collections was 4.8 mm BL (n = 66; range = 2.7–7.5 mm). King mackerel larvae were significantly less common in fall bongo samples in the study area than Gulfwide but occurred at comparable frequencies in fall neuston samples in the 2 areas (Table 3). Relative abundances in the 2 areas differed by < 0.1% (Table 4).

Scomberomorus maculates (39 occurrences; 144 larvae; Figure 57)

Spanish mackerel larvae were also taken as often in bongo as in neuston samples, with the latter gear capturing the majority of specimens. Howver, unlike king mackerel, Spanish mackerel larvae were found in the UNIS study area during spring surveys (Table 1). Mean size in bongo samples was 2.9 mm BL (n = 25; range = 1.7–6.1 mm) and mean size in neuston collections was 7.7 mm BL (n = 74; range = 3.0–15.6 mm). Spanish mackerel larvae were not as evenly distributed

over the study area as king mackerel larvae were during fall surveys (Figure 57). Larvae were significantly more common in the UNIS than Gulfwide survey area during spring surveys but were significantly less common in the UNIS study area during fall surveys (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

Thunnus spp. (165 occurrences; 712 larvae; Figure 58)

Larvae of this genus are difficult to identify. Due to the economic importance of Atlantic bluefin tuna, *Thunnus thynnus*, all tuna larvae captured during SEAMAP spring surveys and initially identified in Poland are re–examined and their identification verified (W. J. Richards, NMFS, pers. comm.). No attempt was made to identify *Thunnus* larvae captured in fall surveys to species. Although far more numerous in neuston samples, *Thunnus* spp. larvae were taken as often in neuston as in bongo samples (Table 1). Mean size in bongo samples was 3.3 mm BL (n = 127; range = 1.8–7.6 mm) and mean size in neuston collections was 5.0 mm BL (n = 312; range = 2.8–10.3 mm). Occurrence and abundance were higher during fall than spring surveys. Tuna larvae were only taken beyond the 200 m isobath during spring surveys but were found from the 200 to within the 50 m isobaths during fall surveys (Figure 58). Frequency of occurrence and relative abundance of tuna larvae within the Gulfwide and UNIS survey areas varied with survey timeframe (Table 3). *Thunnus* spp. larvae were significantly less common in the UNIS study area than Gulfwide survey area during spring surveys and in fall neuston samples but occurred at comparable frequencies in fall bongo samples in the two areas (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

Thunnus thynnus (26 occurrences; 136 larvae; Figure 59)

Atlantic bluefin tuna are managed through international treaties governing their conservation. Annual estimates of larval abundance from SEAMAP spring plankton surveys have been used in stock assessments for this species since the 1980's (Scott et al. 1993). Atlantic bluefin tuna larvae were more frequently taken in neuston than in bongo net samples and were only captured during spring surveys (Table 1). Mean size in bongo samples was 3.9 mm BL (n = 10; range = 2.2–5.5 mm);
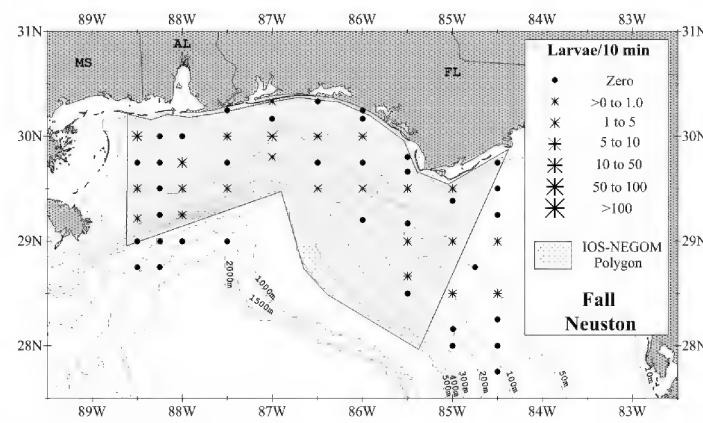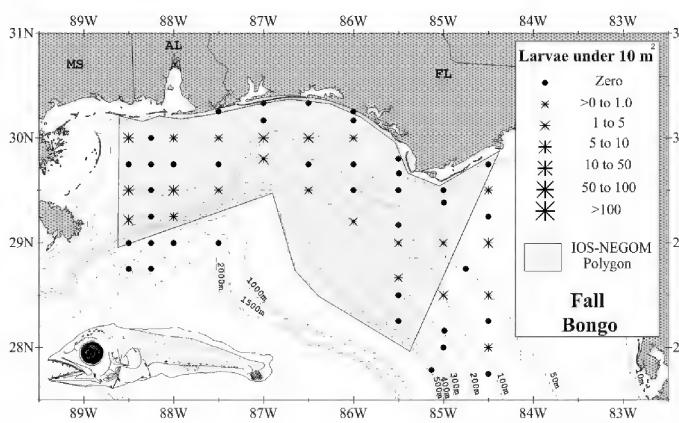
Figure 56. *Mean abundance and mean CPUE of king mackerel,* Scomberomorus cavalla, *larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.*
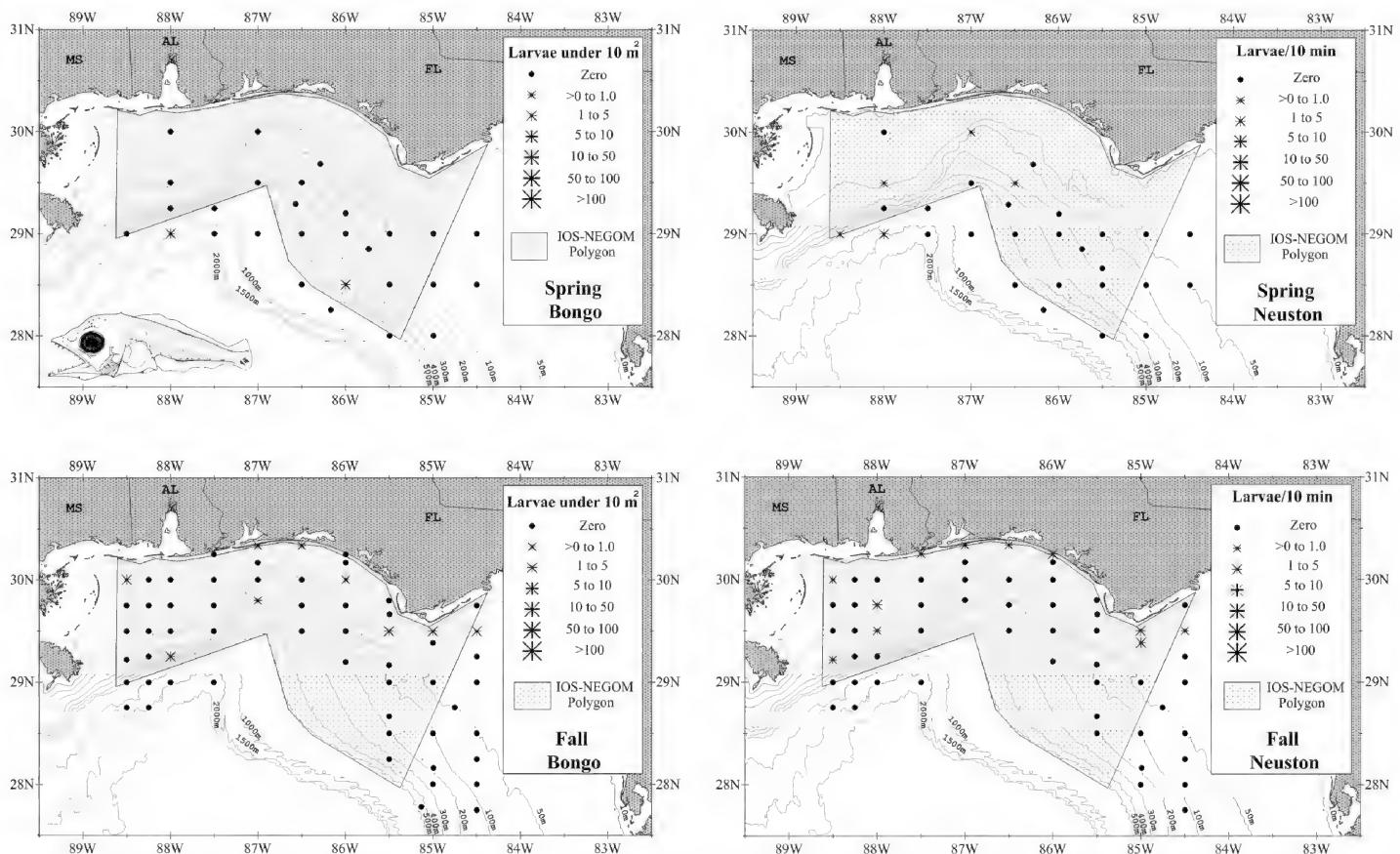
Figure 57. Mean abundance and mean CPUE of Spanish mackerel, Scomberomorus maculatus, larvae at stations in the UNIS study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
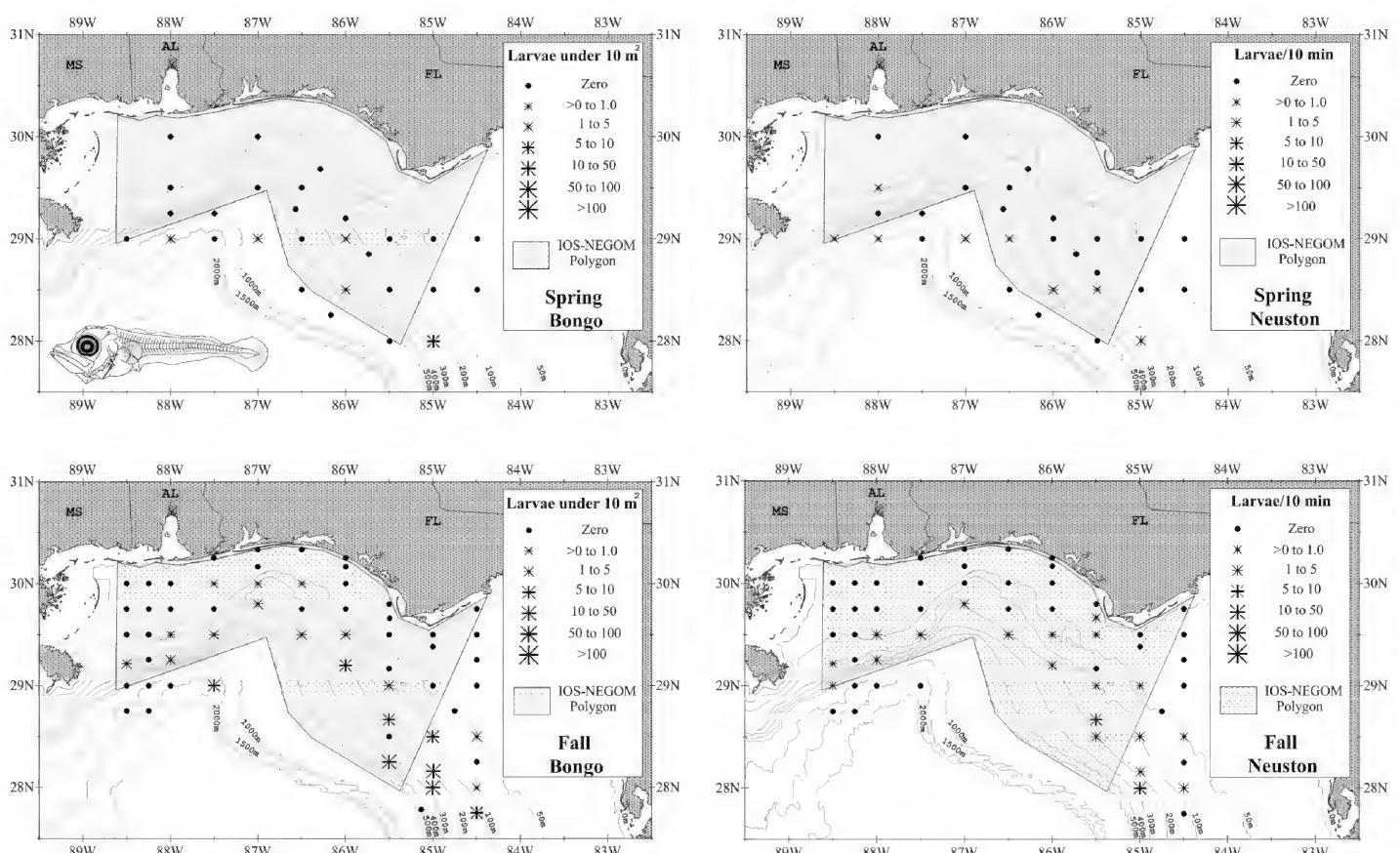

**Figure 58.** Mean abundance and mean CPUE of tuna (Thunnus spp.) larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.

mean size in neuston collections was 4.9 mm BL (n = 88; range = 3.1–6.0 mm). Larvae were found across the study area but mean abundances were highest at stations in the south-eastern corner of the study area (Figure 59). Atlantic bluefin tuna larvae were significantly less common in the UNIS study area than Gulfwide survey area (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

XIPHIIDAE

Xiphias gladius (3 occurrences; 4 larvae; Figure 54)

Swordfish larvae were rare in the UNIS study area. All 4 specimens captured were taken in neuston samples during spring surveys at or beyond the continental shelf–slope break, i.e., beyond the 200 m isobath (Table 1; Figure 54). Overall size range of larvae was 19–46 mm BL. Larvae were relatively more common in Gulfwide collections. They were captured in both bongo and neuston samples during spring surveys and in neuston samples during fall surveys (Table 3). Swordfish larvae were significantly less common in the UNIS study area than Gulfwide survey area (Table 3). Relative abundances and CPUEs in the 2 areas differed by < 0.1% (Table 4).

ISTIOPHORIDAE (38 occurrences; 78 larvae; Figure 60)

Billfish larvae are exceedingly difficult to identify even to
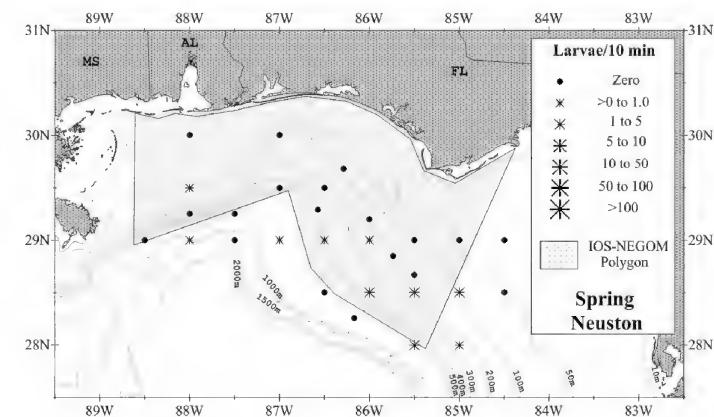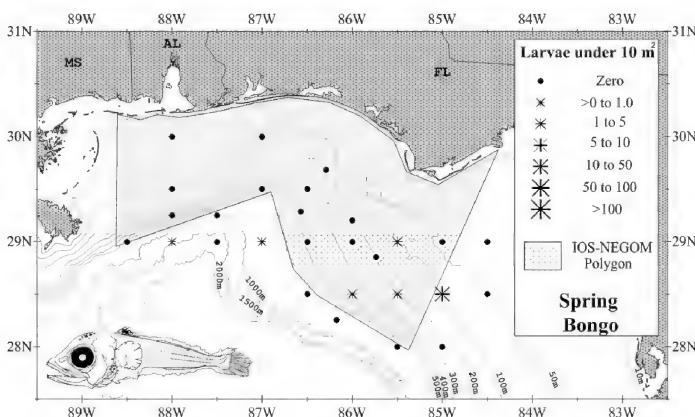
Figure 59. Mean abundance and mean CPUE of bluefin tuna, Thunnus thynnus, larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.
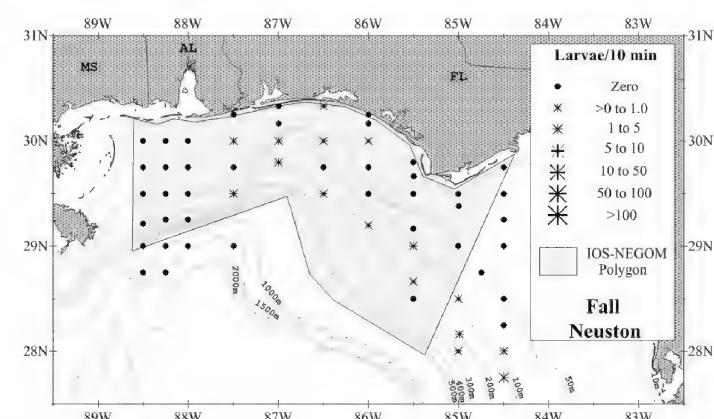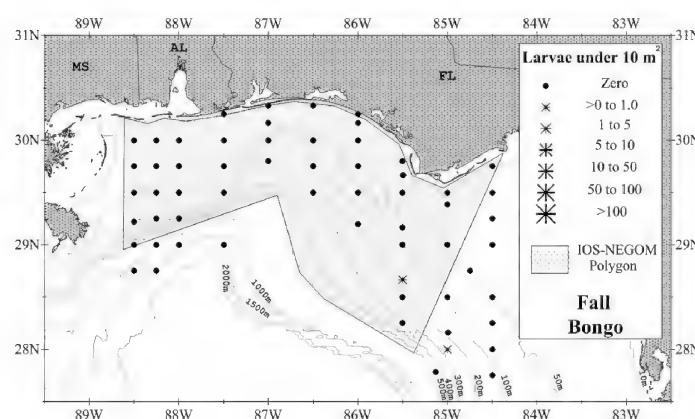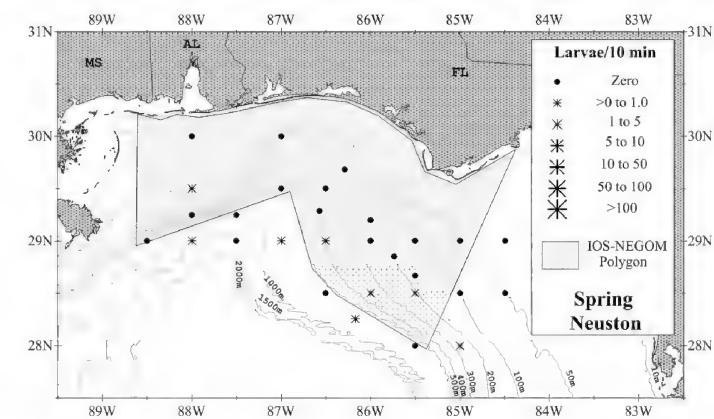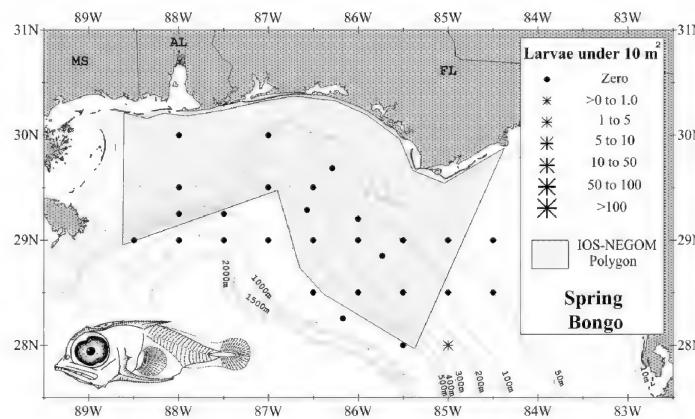
Figure 60. Mean abundance and mean CPUE of billfish (Istiophoridae) larvae at stations in the UNIS study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.

90

the genus level; however, their identity as billfish is indisputable. Recent advances in the application of molecular genetics techniques has made species identifications of billfish larvae more feasible (Luthy et. al. 2005). Most billfish young caught in the UNIS study area were taken in neuston samples with 67% of the specimens being taken during fall surveys (Table 1). Mean size in bongo samples was 3.2 mm BL (n = 5; range = 2.6–4.4 mm) and mean size in neuston collections was 6.2 mm BL (n = 35; range = 3.1–32.8 mm). There was a clear shift in the distribution of billfish larvae from offshore in the spring to more inshore during the fall survey (Figure 60). Larvae were taken most consistently during fall surveys over the 'head' and the eastern slope of the DeSoto Canyon. No billfish larvae were captured west of longitude 87.5°W during fall surveys. There was no statistical difference between the UNIS and Gulfwide survey areas in frequency of occurrence of billfish larvae in spring and fall bongo samples but larvae were taken significantly less frequently in the study area than Gulfwide in spring and fall neuston samples (Table 3). Relative abundance and CPUE in the 2 areas differed by < 0.1% (Table 4).

STROMATEIDAE

Peprilus alepidotus (51 occurrences; 181 larvae; Figure 61)

Harvestfish young, like most other members of the family Stromateidae, are often found concentrated around and associated with floating debris and/or pelagic coelenterates. Larvae were taken primarily in bongo samples and almost exclusively taken during fall surveys (Table 1). Mean size in bongo samples was 2.3 mm BL (n = 129; range = 1.2–11.0 mm) and mean size in neuston collections was 8.9 mm BL (n = 55; range = 2.8–39 mm). Larvae were found throughout the UNIS study area generally within the 100 m isobath with highest mean abundances in bongo samples observed off Cape San Blas inshore of the 50 m isobath off northern Florida (Figure 61). There was a single, spring occurrence in a bongo sample (not shown on Figure 61) at station B153 between the 200 and 300 m contours southwest of the Florida Middle Grounds. There was no statistical difference between the UNIS and Gulfwide survey areas in frequency of occurrence of young harvestfish larvae in spring bongo samples and fall neuston samples, but larvae were taken significantly less often in the study area than Gulfwide in fall bongo samples (Table 3). Relative abundance and CPUE in the 2 areas differed by < 0.1% (Table 4).

Peprilus burti (115 occurrences; 813 larvae; Figure 62)

The Gulf butterfish is a demersal, vertically migrating middle to outer continental shelf species (Vecchione 1987, Herron et al. 1989). Their larvae were more abundant in the UNIS study area than harvestfish larvae. Like its congener, Gulf butterfish larvae were more common in bongo than neuston samples, and were taken almost exclusively during fall surveys (Table 1). Mean size in bongo samples was 2.4 mm BL (n = 338; range = 1.2–11 mm) and mean size in neuston collections was 8.7 mm BL (n = 47; range = 2.5–22.5 mm). Although larvae occurred throughout the study area, highest mean abundances were observed at stations between the 50 and 200 m isobaths along the contours outlining the DeSoto Canyon (Figure 62). Frequency of occurrence of Gulf butterfish larvae in bongo samples was higher in the UNIS study area (26%) than Gulfwide (11%; Table 3). There was no statistical difference between the UNIS and Gulfwide survey areas in frequency of occurrence of young Gulf butterfish larvae in spring bongo samples but larvae were taken significantly more often in the study area than Gulfwide in spring neuston and fall survey samples (Table 3). Relative abundance and CPUE in the 2 areas differed by < 0.5% (Table 4).

Summary of Distributional Observations

A complete representation of the seasonality of ichthyoplankton occurrence and abundance in the NEGOM cannot be produced from these SEAMAP data since only 2 survey time frames yielded data for this synopsis. However, these
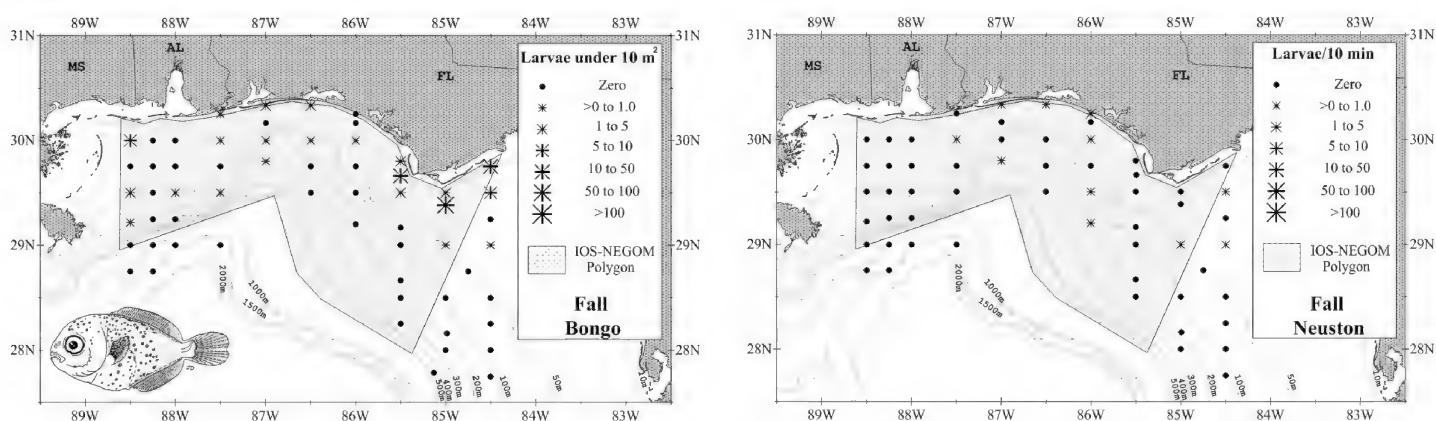

Figure 61. *Mean abundance and mean CPUE of harvestfish,* Peprilus alepidotus, *larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. (Map of one occurrence in spring survey bongo samples is not shown.) UNIS and SEAMAP defined in Figure 1.*
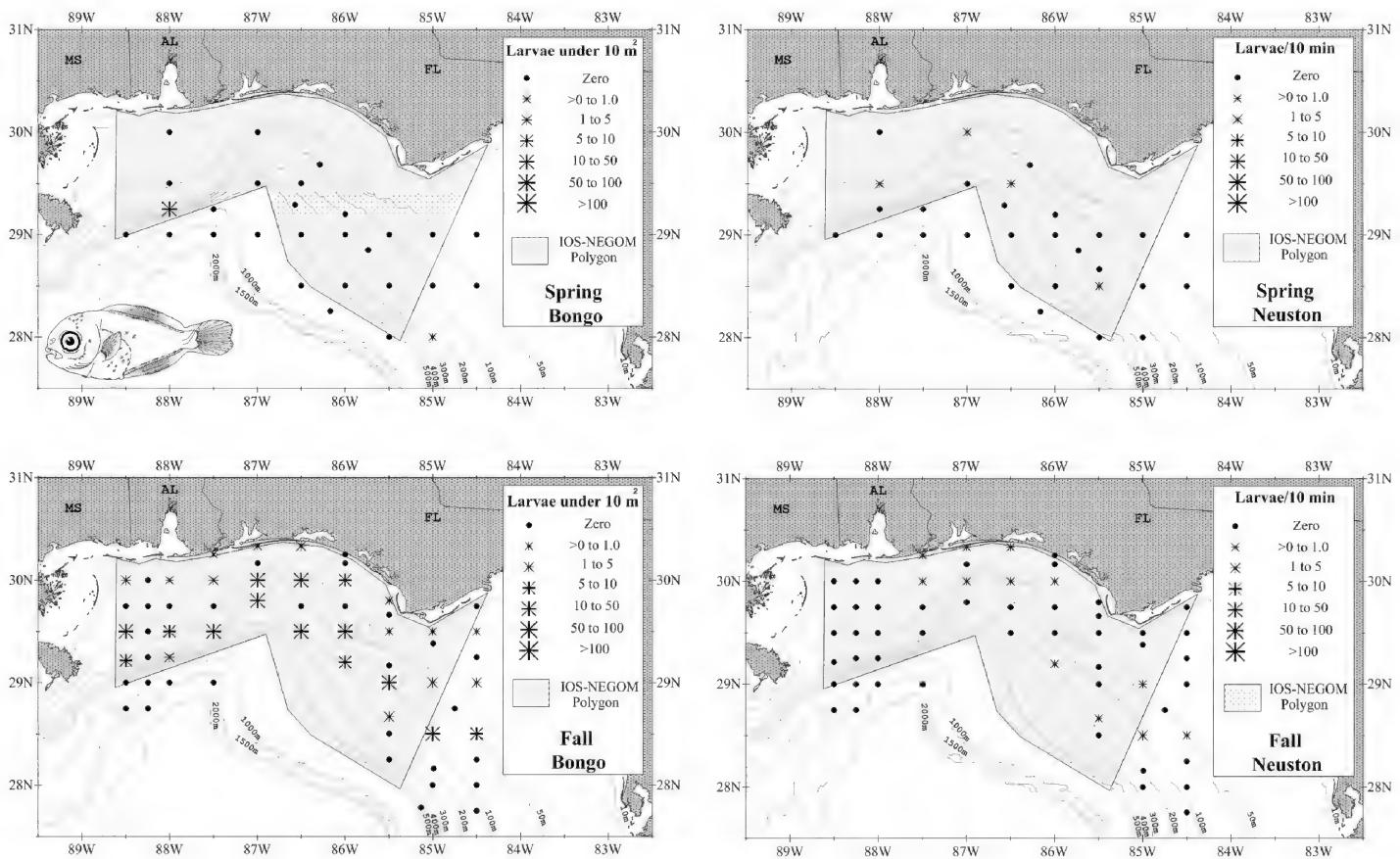
Figure 62. Mean abundance and mean CPUE of Gulf butterfish, Peprilus burti, larvae at stations in the UNIS Study area captured during SEAMAP surveys, 1982-1999. UNIS and SEAMAP defined in Figure 1.

2 periods, principally mid—April through May and September to mid—October, encompass the spawning seasons and peak occurrence of the majority of shelf dwelling species in the GOM (Ditty 1986; Ditty et al. 1988). Houde and Chitty (1976) found ichthyoplankton abundance to be highest on the west Florida shelf during May through September. Of course, depending on specific taxa of interest there are notable exceptions to this, namely, late fall to winter spawning species such as most of the groupers, tilefishes, porgies, menhaden and striped mullet.

Three coarse measures of fish spawning, ichthyoplankton abundance, and zooplankton abundance indicated that the UNIS study area contributed more fish eggs, total larvae and zooplankton to Gulfwide survey totals than would be expected from the number of samples taken in that region. This was more evident during spring than fall surveys, and is probably related to the close proximity of UNIS survey stations to the Mississippi River and the inshore penetration of the DeSoto Canyon. The vast majority of spring survey stations outside the study area are located in open GOM waters beyond the influence of nutrient enriched, continental shelf waters.

During spring surveys, total ichthyoplankton abundance both in the water column and at the surface was dispropor-

tionately higher (5–9%) in the study area than Gulfwide, while during fall surveys the reverse was true (2–6% lower than expected). This latter finding is probably related to the reduced amount of shelf area in the UNIS study area compared to the remainder of the SEAMAP fall survey area especially in the western GOM. Zooplankton biomass was also greater in the study area relative to the entire survey area during both survey time frames but the difference was greater (8% above expectation) during fall surveys.

Statistical comparison of the frequency of occurrence of the 61 ichthyoplankton taxa selected for this study revealed that the larvae of many were taken significantly more often in the UNIS area than expected based on their occurrence Gulfwide. In contrast, gross measures of relative ichthyoplankton abundance and CPUE, with a few notable exceptions, indicated little difference between the study areas. Thirteen of these taxa were taken significantly more often in the study area during the season and collecting gear combination that accounted for the highest catches. These taxa were: Muraenidae, *E. teres*, Engraulidae, Sternoptychidae, Synodontidae, *C. bermudensis*, Serranidae, *Decapterus* spp., *Seriola* spp., *R. aurorubens*, Mullidae, Labridae, and *P. burti*. The relative abundance and CPUE in the UNIS study area of only the Engraulidae and Mullidae accounted for a dis-

proportionately higher than expected value of the percent of abundance of total fish larvae captured Gulfwide.

The 13 taxa includes fishes from mesopelagic, coastal and shelf demersal and pelagic, and reef assemblages indicating that the NEGOM is an important spawning and/or nursery area for a diverse group of fishes. This diversity reflects the wide variety of available habitats in this region of the GOM that range from shallow mud to sand to deep hard/live bottoms all adjoining deep oceanic waters. The occurrence and relative abundance of numerous other taxa in the study area were similar to Gulfwide values. Therefore, as the larvae of more species are identified in SEAMAP collections using updated descriptive information, the number of taxa for which the NEGOM proves to be an important spawning and/or nursery area will undoubtedly increase.

The importance of the NEGOM to production of many economically valuable fishes including coastal pelagic, reef and highly migratory taxa is supported by the consistent occurrence of their young in the UNIS study area. Amberjack (*Seriola* spp.) and vermilion snapper (*R. aurorubens*) larvae were relatively more common in the study area than Gulfwide during the season of their highest abundance. At other times, some larvae were found to be more common in the study area than Gulfwide at a time outside of the documented spawning season, such as Spanish mackerel (*S. maculatus*) during spring surveys. This latter finding may indicate that Spanish mackerel spawning begins earlier in the NEGOM than in other areas of the northern GOM. Red snapper (*L. campechanus*) larvae were in general less common in the study area than Gulfwide, but their relative abundance in study area neuston samples was similar to the Gulfwide value. Young dolphins (Coryphaenidae) were less common in the study area during spring (offshore) surveys when they were most abundant but were as common in the UNIS area as Gulfwide during fall surveys over the continental shelf. Occurrences of billfish (Istiophoridae), wahoo (*A. solandri*), and cobia (*R. canadum*) larvae in plankton collections are rare events anywhere. Thus, the consistent occurrence of these species in SEAMAP collections in the UNIS study area is noteworthy, indicating that these highly migratory fishes spawn in the NEGOM region.

Several general distribution patterns emerged from an examination of the occurrences of larvae within the UNIS study area. Of the 61 selected taxa, the larvae of 4 taxa occurred predominately west of 87°W longitude; *R. canadum*, *Caranx* spp., *L. campechanus*, and *T. lepturus*. The larvae of 14 taxa occurred mostly at stations east of the 87th meridian; *S. aurita*, *E. teres*, *C. bermudensis*, Epinephelinae, Grammistinae, Priacanthidae, *Seriola* spp., *L. griseus*, Scaridae, Istiophoridae, Apogonidae, Haemulidae, Chaetodontidae, and Pomacanthidae. A number of taxa in this latter group were found predominately east of longitude 86.5°W, including L. griseus, Haemulidae, Chaetodontidae, and Pomacanthidae.

These patterns coincide with distinct changes in topography, bottom type and hydrography of the region and, in turn, available habitats and associated biological communities. The northern rim of DeSoto Canyon cuts into the inner continental shelf to a minimum depth of 50–60 m dividing the NEGOM shelf into distinct western and eastern sectors subject to different physical and biological influences. Oceanographically, cold deep water, driven by the GOM Loop Current (Maul 1977), rides up the canyon impinging upon the inner shelf (Müller–Karger et al. 2001). The area west of 87°W consists of a broad predominately mud and clay (terrigenous sediments) covered shelf that adjoins several extensive estuarine systems and can be influenced directly by the Mississippi River. East of that meridian the shelf narrows, sand and carbonate sediments are dominant and riverine influence is minimal.

The influence of the DeSoto Canyon on the fish fauna of the NEGOM is also profound, differentiating both the demersal (Weaver et al. 2002) and pelagic fish faunas, including current–borne ichthyoplankton. The SEAMAP distribution patterns for larvae of 6 taxa clearly coincide with the 50–500 isobaths outlining the submarine canyon: *C. bermudensis*, Sternoptychidae, Paralepididae, Anthiine, and *P. aquilonaris*. The distribution of larval *P. burti* was also linked to the canyon, but deviated somewhat from this pattern. Highest mean abundances were consistently located over the canyon, although Gulf butterfish larvae also occurred inshore of the canyon.

Although discrete depth sampling was not conducted during SEAMAP surveys, the 2 types of plankton nets employed provided samples from distinct and separate segments of the water column. The neuston net sampled the upper half–meter of the ocean surface layer. The bongo net sampled the entire water column from sub–surface to near bottom (or to a maximum depth of 200 m when bottom depth was greater). Contrasting the catches of the 2 gear types provided some insights into utilization of 2 different oceanic regimes by fish larvae in the study area. The young of 11 taxa, including highly migratory, pelagic, and reef fishes, were found predominantly in the surface layer of the ocean: *X. gladius*, Istiophoridae, *T. thynnus*, *R. canadum*, *Caranx* spp., *Seriola* spp., Coryphaenidae, *L. surinamensis*, Muraenidae, Holocentridae, and Mullidae. For these taxa, over 85% of specimens were taken in surface waters, and over 70% of captures occurred in surface waters. The young of *X. gladius*, *R. canadum*, and *L. surinamensis* were never captured below the surface layer (i.e., never in bongo nets). All remaining taxa considered in this study were as numerous, or more numerous, below the surface layer (i.e., in bongo net collections) as at the surface (i.e., in neuston net collections). Among the young of hard–bottom and deep–reef fishes analyzed from study area collections, 6 were found principally below the surface layer, occurring in over 70% of bongo samples: *C. bermudensis*, An-

thiinae, Epinephelinae, Haemulidae, Labridae and Scaridae. Except for the Anthiinae, over 90% of specimens in these taxa were taken in the water column. In the Anthiinae, 69% of specimens were taken in the water column. The young of 3 additional hard/live bottom taxa; Priacanthidae, Pomacentridae and Acanthuridae, occurred with equal frequency in both surface and water column collections.

Limited size data were summarized for the young of taxa representing fishes living in or near hard/live bottom habitats, namely the 4 subfamilies of sea basses (Serranidae) and 10 families of obligate reef fishes. Due to the difficulties inherent in these data (i.e., not all larvae in collections were measured), only the incidence of the largest and smallest specimens relative to position in the water column were examined. There seemed to be a difference in the size of larvae captured in the surface layer and throughout the water for some reef taxa. The largest individuals of 3 taxa were consistently taken in neuston samples. This was most evident among young Holocentridae but was also true for the Priacanthidae and Pomacentridae. The early life histories of the first 2 families are known to include a pelagic juvenile stage of long duration prior to settlement (Thresher 1984). Early life stages of representatives of all 3 families, especially the pomacentrid, *Abudefduf saxatilis*, are consistently taken in floating *Sargassum* in the western Atlantic Ocean and NEGOM (Dooley 1972, Bortone et al. 1977, Moser et al. 1998, Franks et al. 2002). The size distributions of the most ubiquitous and numerous reef fishes in UNIS study area plankton collections, the Labridae and Scaridae, were essentially the same in both surface and water column collections. Among sea bass larvae, larger anthiines were taken in water column samples while individuals in the largest size classes of the Serranine and Grammistinae were equally represented in the 2 sampled segments of the water column.

The consistent presence of fish eggs throughout the study area at mean abundances > 100 under 10 m² sea surface indicates that the NEGOM is an important spawning area. Additional evidence of high spawning activity in the region comes from a survey of the entire west Florida shelf (Houde and Chitty 1976). These workers found that the most intense spawning of fishes occurred north of latitude 27°15′N, i.e., the area adjoining the UNIS study area to the east. The presence of larvae in the 1.5 and 2.0 mm size classes is further evidence of local spawning. Small sea bass larvae in those size classes were collected in the UNIS study area indicating, unambiguously, that these fishes spawn in the NEGOM region. Small (≤ 2.0 mm) larvae of 7 of the 10 selected reef fish families were also present in study area samples. The smallest larvae of the Chaetodontidae, Pomacanthidae, and Acanthuridae taken in UNIS study area samples were 2.5, 3.1, and 3.5 mm BL, respectively. Despite the small number of specimens captured, it is more than likely based on the known areas of hard bottom habitat in the NEGOM that

these taxa also spawn in this area.

Local spawning, however, may not be the only source of reef fish larvae in NEGOM waters. The Loop Current and its associated eddies and rings are known to exert the dominant dynamic influence not only in the open GOM but also on the continental shelf and slope and facilitate exchanges of water mass between them (Maul 1977, Vukovich and Crissman 1986, Kelly 1991, Hamilton 1992, Berger et al. 1996, Nowlin et al. 1998). The UNIS study area is consistently in the direct path of the Loop Current; it has been shown that the shelf edge region off Mississippi and Alabama is influenced by the Loop Current 40% of the time (Kelly 1991). Additionally, pools of Loop Current water formed by short–lived rings can intrude into the UNIS study area at least as often as every 2 years (Muller–Karger et al. 2001). It is probable, therefore, that the early life stages of hard/live bottom fishes are periodically transported into the study area via Loop Current intrusions, providing an extrinsic source of recruitment. However, larvae produced in the NEGOM may likewise be either retained there or exported to other GOM reefs via the same mechanisms. Hanisko and Lyczkowski–Shultz (2003) examined the distribution of labrid and scarid larvae from SEAMAP collections Gulfwide in light of the Loop Current and its associated eddies and rings. These authors suggested that, depending on species–specific, planktonic stage durations, larvae produced on reefs throughout the northern GOM could be entrained in currents produced by Loop Current eddies and could return in time to settle on their natal reefs or, alternatively, could be exported to settle on distant GOM reefs.

This synopsis represents an examination of the most extensive set of ichthyoplankton data available for the northern GOM, namely data generated from SEAMAP plankton surveys ongoing since 1982. The specific purpose of this analysis of historical SEAMAP ichthyoplankton data was to characterize occurrence and relative abundance of young fishes in the northeastern region of the GOM and to examine the region's relative contribution to the early life histories of fishes as compared to the entire GOM within the U.S. EEZ. This summary has revealed that the NEGOM should be considered an important if not critical habitat for the young of a diverse, perhaps even unique, assemblage of fish larvae. The varied and juxtaposed essential fish habitats of the NEGOM result in an area that is used as spawning and nursery grounds for estuarine and coastal, hard/live bottom, soft bottom, and oceanic fishes.

Future Directions

Since this data summary was first produced as USGS Project report USGS SIR–2004–5059 (http://cars.er.usgs.gov/coastaleco/), a significant contribution to larval fish identifications in the region has been published and plankton sampling during SEAMAP surveys has been expanded; both of which address shortcomings of historical SEAMAP

ichthyoplankton surveys and data as revealed here. The recently published guide to the early life stages of marine fishes of the western central Atlantic Ocean (Richards 2006) brings together in a single work all previously published, as well as new larval descriptions. This compilation will facilitate the re—examination and more precise identification (i.e., to lower taxonomic levels) of archived SEAMAP specimens. Improved taxonomic resolution will allow SEAMAP ichthyoplankton data to be used to describe critical spawning and nursery habitats, relationships between oceanographic processes and pre—settlement stage larvae, or to reveal recruitment dynamics and the effects of perturbations to the environment for more species than was previously possible.

Additional shortcomings of the earlier SEAMAP ichthyoplankton surveys lie in the realm of seasonal coverage, discrete depth, and directed sampling. The original plan to sample throughout the GOM in all seasons has never been realized. As a result, there are major gaps in data and information for species that spawn in areas and at times that remain un—surveyed. The most egregious deficiency is the lack of information on winter spawning species. In order to fill this data gap, NMFS began a biannual, SEAMAP winter plankton survey in 2007 which over time will result in a more comprehensive set of ichthyoplankton data for all marine fishes in the GOM. Also starting with the 2007 winter survey, NMFS began taking discrete depth ichthyoplankton collections with a 1 m multiple opening and closing net and environmental sensing system (MOCNESS) during SEAMAP plankton surveys. Position in the water column can have a direct influence on dispersal of fish larvae (Lyczkowski—Shultz and Steen 1991) and the pre—settlement and pelagic juvenile stages of many reef fishes are capable of adjusting their vertical position in the water column; some have been shown to maintain a preferred depth (Leis et al. 1996, Cowan and Sponaugle 1997). Subsurface currents may be an important mechanism for either the retention of larvae near or their transport to the habitats and communities where they will eventually settle and take up demersal existence.

Implementation of plankton sampling targeting oceanographic features of interest began in 2008 during the spring SEAMAP survey. Satellite imagery was used to direct additional, 'off the grid' sampling in the vicinity of Loop Current eddies and convergence zones where it is believed that Atlantic bluefin tuna spawn and where their larvae may be concentrated (Muhling et al. 2010). Ichthyoplankton sampling in the GOM relative to Loop Current fronts and associated convergence zones has shown that the larvae of tunas, wrasses and parrotfishes are more abundant in areas dominated by these oceanographic features (Richards et al. 1989, Hanisko and Lyczkowski—Shultz 2003).

Ultimately the data from more specialized sampling of this kind will reveal in more detail the coupling between oceanographic processes and recruitment of not only reef species but also the young of fishes such as mullet and menhaden that are spawned in open GOM waters and must return to nearshore habitats (Richards and Lindeman 1987). This new, ever evolving SEAMAP ichthyoplankton database will allow future researchers to conduct more sophisticated analyses of larval fish assemblages and provide more detailed insights into the early life histories of marine fishes in the GOM ecosystem.

LITERATURE CITED

Ahlstrom, E.H. and H.G. Moser. 1981. Systematics and development of early life history stages of marine fishes: achievements during the past century, present status and suggestions for the future. Rapports et Procès—Verbaux des Rèunions Conseil International pour L'Exploration de la Mer, Copehagen, Demark. 178:541—546.

Berger, T.J., P. Hamilton, J.J. Singer, and R.R. Leben. 1996. Louisiana/Texas Shelf Physical Oceanography Program: eddy circulation study, final synthesis report. OCS Study MMS 96—0051. United States Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA, USA, 324 p. (available at: http://www.data.boem.

gov/PI/PDFImages/ESPIS/3/3289.pdf)

Bortone, S.A., P.A. Hastings, and S.B. Collard. 1977. The pelagic—*Sargassum* ichthyofauna of the eastern Gulf of Mexico. Northeast Gulf Science 1:60–67.

Cowan, R.K. and S. Sponaugle. 1997. Relationships between early life history traits and recruitment among coral reef fishes. In: C. Chambers and E. A. Trippel, eds. Early Life History and Recruitment in Fish Populations. Chapman and Hall, London, England, p. 423–449.

Ditty, J.G. 1986. Ichthyoplankton in neritic waters of the northern Gulf of Mexico off Louisiana: composition, relative abundance, and seasonality. Fishery Bulletin 84:935–946.

Ditty, J.G. and R.F. Shaw. 1996. Spatial and temporal distribution of larval striped mullet (*Mugil cephalus*) and white mullet (*M. curema*, Family: Mugilidae) in the northern Gulf of Mexico, with notes on mountain mullet, *Agonostomus monticola*. Bulletin of Marine Science 59:271–288.

Ditty, J.G., G.C. Zieske, and R.F. Shaw. 1988. Seasonality and depth distribution of larval fishes in the northern Gulf of Mexico above latitude 26°00 N. Fishery Bulletin 86:811–823.

Dooley, J.K. 1972. Fishes associated with the pelagic *Sargassum* complex, with discussion of the *Sargassum* community. Contributions in Marine Science 16:1–32.

Farooqi, T.W., J.G. Ditty, and R.F. Shaw. 2006. Engraulidae: Anchovies. In: W.J. Richards, ed. Early Stages of Atlantic Fishes – An Identification Guide for the Western Central North Atlantic, Vol. II, CRC Press, Boca Raton, FL, USA, p. 101–127.

Franks, J.S., B.R. Comyns, J.R. Hendon, E.R. Hoffmayer, R.S. Waller, N.M. Crochet, and M.E. Blake. 2002. Investigation of juvenile fishes that utilize *Sargassum* and frontal zones as essential habitat in Mississippi waters and adjacent Gulf waters. Contact No. 067–C–Sargassum. Final Report. U.S. Fish and Wildlife Service, Atlanta, GA, USA, 49 p.

Gardner, J.V., P. Dartnell, K.J. Sulak, B.Calder, and L. Hellequin. 2001. Physiography and late Quaternary—Holocene Processes of northeastern Gulf of Mexico outer continental shelf off Mississippi and Alabama. Gulf of Mexico Science 19:132–157.

Hanisko, D.S. and J. Lyczkowski–Shultz. 2003. Occurrence and pelagic habitat of reef fish larvae in the Gulf of Mexico. In: D.R. Stanley and A. Scarborough—Bull, eds. Fisheries, Reefs, and Offshore Development. American Fisheries Society, Symposium 36, Bethesda, MD, USA, p. 205–224.

Hamilton, P. 1992. Lower continental slope cyclonic eddies in the central Gulf of Mexico. Journal of Geophysical Research 97:2185–2200.

Herron, R.C., T.D. Leming, and J. Li. 1989. Satellite—detected fronts and butterfish aggregations in the northeastern Gulf of Mexico. Continental Shelf Science 9:569–588.

Hoese, H.D. and R.H. Moore 1977. Fishes of the Gulf of Mexico – Texas, Louisiana, and Adjacent Waters. Texas A&M University Press, College Station, TX, USA, 327 p.

Houde, E.D. and N.Chitty. 1976. Seasonal abundance and distribution of zooplankton, fish eggs, and fish larvae in the eastern Gulf of Mexico, 1972–74. NMFS SSRF–701. Technical Report. National Oceanic and Atmospheric Administration, Seattle, WA, USA, 18 p.

Houde, E.D., J.G. Leak, C.E. Dowd, S.A. Berkeley, and W.J. Richards. 1979. Ichthyoplankton abundance and diversity in the eastern Gulf of Mexico. Contract AA550–CT7–28. NTIS PB–299839. Final Report, Bureau of Land Management, Alexandria, VA, USA, 546 p.

Kavanagh, K.D., J.M. Leis, and D.S. Rennis. 2000. Pomacentridae (Damselfishes). In: J.M. Leis and B.M. Carson–Ewart, eds. The Larvae of Indo–Pacific Coastal fishes: An Identification Guide to Marine Fish Larvae. Fauna Malesiana Handbook 2, Brill, The Netherlands, p. 526–535.

Kelley, S. 2006. Chaetodontidae: butterflyfishes. In: W.J. Richards, ed. Early Stages of Atlantic Fishes – An Identification Guide for the Western Central North Atlantic, Vol. II, CRC Press, Boca Raton, FL, USA, p. 1767–1786.

Kelly, F.J. 1991. Physical oceanography. In: J.M. Brooks and C.P. Giammona, eds. Mississippi–Alabama Continental Shelf Ecosystem Study Data Summary and Synthesis. OCS Study MMS 91–0062. United States Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA, USA, 42 p. (available at http://www.data.boem.gov/PI/PDFImages/ESPIS/3/3645.pdf)

Kendall, A.W., Jr. and A.C. Matarese. 1994. Status of early life history descriptions of marine teleosts. Fishery Bulletin 92:725–736.

Kramer, D., M.J. Kalin, E.G. Stevens, J.R. Thrailkill, and J.R. Zweifel. 1972. Collecting and processing data on fish eggs and larvae in the California Current region. NMFS Circular 370. Technical Report, National Oceanic and Atmospheric Administration, Seattle, WA, USA, 38 p.

Laroche, W.A., J.G. Ditty, J.T. Lamkin, and S.R. Whitcraft. 2006. Carangidae: Jacks. In: W.J. Richards, ed. Early Stages of Atlantic Fishes – An Identification Guide for the Western Central North Atlantic, Vol. II, CRC Press, Boca Raton, FL, USA, p. 1439–1509.

Leis, J.M. 1989. Larval biology of butterflyfishes (Pisces, Chaetodontidae): what do we really know? Environmental Biology of Fishes 25:87–100.

Leis, J.M. and D.S. Rennis. 1983. The larvae of Indo–Pacific coral reef fishes. New South Wales University Press, Sydney, Australia and University of Hawaii Press, Honolulu, HI, USA, 269 p.

Leis, J.M., H.P.A. Sweatman, and S.E. Reader. 1996. What the pelagic stages of coral reef fishes are doing out in blue waters: daytime field observations of larval behavioural capabilities. Marine and Freshwater Research 47:401–411.

Lindeman, K.C. and W.J. Richards. 2006. Haemulidae: Grunts. In: W.J. Richards, ed. Early Stages of Atlantic Fishes – An Identification Guide for the Western Central North Atlantic, Vol. II, CRC Press, Boca Raton, FL, USA, p. 1597–1646.

Lindeman, K.C., T.N. Lee, W. D. Wilson, R. Claro, and J. S. Ault. 2001. Transport of larvae originating in southwest Cuba and the Dry Tortugas: evidence for partial retention in grunts and snappers. Proceedings of the Gulf and Caribbean Fisheries Institute 52:732–747.

Lindeman, K.C., W.J. Richards, J. Lyczkowski–Shultz, D.M. Drass, C.B. Paris, J.M. Leis, M. Lara, B.H. Comyns. 2006. Lutjanidae: Snappers. In: W.J. Richards, ed. Early Stages of Atlantic Fishes – An Identification Guide for the Western Central North Atlantic, Vol. II, CRC Press, Boca Raton, FL, USA, p. 1549–1586.

Luthy, S.A., R.K. Cowen, J.E. Serafy, and J.R. McDowell. 2005. Toward identification of larval sailfish (*Istiophorus platypterus*), white marlin (*Tetrapturus albidus*), and blue marlin (*Makaira nigricans*) in the western North Atlantic Ocean. Fishery Bulletin 103:588-600.

Lyczkowski–Shultz, J. and D.S. Hanisko. 2007. A time series of observations of red snapper larvae from SEAMAP surveys 1982–2003: seasonal occurrence, distribution, abundance and size. In: W.F. Patterson, III, J. H. Cowan, Jr., G.R. Fitzhugh, and D.L. Nieland, eds. Red Snapper Ecology and Fisheries in the U.S. Gulf of Mexico. American Fisheries Society, Symposium 60, Bethesda, MD, USA, p. 3–23.

Lyczkowski–Shultz, J. and J.P. Steen, Jr. 1991. Diel vertical distribution of red drum *Sciaenops ocellaltus* larvae in the northcentral Gulf of Mexico. Fishery Bulletin 89:631–641.

Lyczkowski–Shultz, J.M. Konieczna, and W.J. Richards. 2000. Occurrence of the larvae of beryciform fishes in the Gulf of Mexico. Bulletin of the Sea Fisheries Institute 3 (151):55–66.

Lyczkowski–Shultz, J., D.S. Hanisko, K.J. Sulak, and G.D. Dennis, III. 2004. Characterization of Ichthyoplankton within the U.S. Geological Survey's Northeastern Gulf of Mexico Study Area —Based on Analysis of Southeast Area Monitoring and Assessment Program (SEAMAP) Sampling Surveys, 1982–1999. USGS SIR–2004–5059. NEGOM Ichthyoplankton Synopsis Final Report. U.S. Department of the Interior, U.S. Geological Survey, Gainesville, FL, USA, 140 p.

Marancik, K., D. Richardson, J. Lyczkowski–Shultz, M. Konieczna and R. Cowen. 2010. Evaluation of morphological characters to identify grouper (Serranidae: Epinephelini) larvae in the Gulf of Mexico using genetically identified specimens. Bulletin of Marine Science 86: 571–624.

Marancik, K., D. Richardson, J. Lyczkowski–Shultz, M. Konieczna and R. Cowen. 2012. Spatial and temporal distribution of grouper larvae (Serranidae: Epinephelini) in the Gulf of Mexico and Straits of Florida, Fishery Bulletin 110:1–20.

McBride, R.S., C.R. Rocha, R.Ruiz–Carus, and B.W. Bowen. 2010. A new species of ladyfish of the genus *Elops* (Elopiformes: Elopidae) from the western Atlantic Ocean. Zootaxa 2346:29–41.

Maul, G.A. 1977. The annual cycle of the Gulf Loop Current, part I: observations during a one–year time series. Journal of Marine Research 35:29–47.

Moser, M.L., P.J. Auster, and J.B. Bichy. 1998. Effects of mat morphology on large *Sargassum*–associated fishes: observations from a remotely operated vehicle (ROV) and free–floating video camcorders. Environmental Biology of Fishes 51:391–398.

Muhling, B.A., J.T. Lamkin, and M.A. Roffer. 2010. Predicting the occurrence of Atlantic bluefin tuna (*Thunnus thynnus*) larvae in the northern Gulf of Mexico: building a classification model from archival data. Fisheries Oceanography 19:526–539.

Muller–Karger, F.E., F. Vukovich, R. Leben, B. Nababan, C. Hu, and D. Myhre. 2001. Surface circulation and the transport of the Loop Current into the northeastern Gulf of Mexico. OCS Study MMS 2001–102. Final Report. United States Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA, USA, 36 p. (available at: http://www.data.boem.gov/PI/PDFImages/ESPIS/3/3150.pdf)

Nowlin, W.D., Jr., A.E. Jochens, R.O. Reid, and S.F. DiMarco. 1998. Texas–Louisiana shelf circulation and transport processes study: synthesis report, volume I. OCS Study MMS 98–0035. Technical report. United States Department of the Interior, Mineral Management Service, Gulf of Mexico OCS Region, New Orleans, LA, USA, 435 p. (available at: http://www.data.boem.gov/PI/PDFImages/ESPIS/3/3233.pdf)

Posgay, J.A. and R.R. Marak. 1980. The MARMAP bongo zooplankton samplers. Journal of Northwest Atlantic Fisheries Science 1:9–99.

Rakocinski, C.F., J. Lyczkowski–Shultz, and S.L. Richardson. 1996. Ichthyoplankton assemblage structure in Mississippi Sound as revealed by canonical correspondence analysis. Estuarine, Coastal and Shelf Science 43:237–257.

Rester, J.K., N. Sanders, Jr., D.S. Hanisko, and B. Pellegrin. 2000. SEAMAP Environmental and Biological Atlas of the Gulf of Mexico, 1998. No. 75. Gulf States Marine Fisheries Commission, Ocean Springs, MS, USA, 243 p.

Richards, W.J. 1990. List of the fishes of the western central Atlantic and the status of early life stage information. NMFS-SE-FC-267. Technical Memorandum. National Oceanographic and Atmospheric Administration, Miami, FL, USA, 88 p.

Richards, W.J. 2006. Early Stages of Atlantic fishes: An Identification Guide for the Western Central North Atlantic, Vols. 1 & 2. CRC Press, Boca Raton, FL, USA. 2,672 p.

Richards, W.J. and K.C. Lindeman. 1987. Recruitment dynamics of reef fishes: planktonic processes, settlement, and demersal ecologies, and fishery analysis. Bulletin of Marine Science 41:392–410.

Richards, W.J., T. Leming, M.F. McGowan, J.T. Lamkin, and S. Kelley–Fraga. 1989. Distribution of fish larvae in relation to hydrographic features of the Loop Current boundary in the Gulf of Mexico. In: J.H.S. Blaxter, J.C. Gamble, and H. v. Westernhagen, eds. The Early Life History of Fish. Rapports et Procès–Verbaux des Rèunions Conseil International pour L'Exploration de la Mer, Copenhagen, Denmark, Conseil International pour l' Exploration de la Mer 191, p. 169–176

Richards, W.J., M.F. McGowan, T. Leming, J.T. Lamkin, and S. Kelley. 1993. Larval fish assemblages at the Loop Current boundary in the Gulf of Mexico. Bulletin of Marine Science 53:475–537.

Richards, W. J., J. Lyczkowski–Shultz, and M. Konieczna. 2006a. Holocentridae: Squirrelfishes. In: W.J. Richards, ed. Early Stages of Atlantic Fishes – An Identification Guide for the Western Central North Atlantic, Vol. II, CRC Press, Boca Raton, FL, USA, p.1047–1050.

Richards, W.J., C.C. Baldwin, and A. Röpke. 2006b. Serranidae: Seabasses. In: W.J. Richards, ed. Early Stages of Atlantic Fishes – An Identification Guide for the Western Central North Atlantic, Vol. II, CRC Press, Boca Raton, FL, USA, p. 1225–1331.

Sale, P.F. 1991. Introduction. In: P.F. Sale, ed. The Ecology of Fishes on Coral Reefs. Academic Press, New York, NY, USA, p. 3–11.

Scott, G.P., S.C. Turner, C.B. Grimes, W.J. Richards, and E.B. Brothers. 1993. Indices of larval bluefin tuna, *Thunnus thynnus*, abundance in the Gulf of Mexico: modeling variability in growth, mortality, and gear selectivity. Bulletin of Marine Science 53:912-929.

Smith, P.E. and S.L. Richardson, eds. 1977. Standard Techniques for Pelagic Fish Egg and Larva Surveys. FAO Fisheries Technical Paper 175, Rome, Italy, 100 p.

Sulak, K.J., D.C. Weaver, and S.W. Ross. 2000. Fish community structure: Mississippi/Alabama Pinnacle Trend. In: M. Mckay and J. Nides, eds. OCS Study MMS 2000–030. Proceedings of the Eighteenth Gulf of Mexico Information Transfer Meeting, Kenner, LA, December 1998, U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA, USA, p. 194–203.

Thresher, R.E. 1984. Reproduction in reef fishes. T.F.H. Publications, Inc., Neptune City, NJ, USA, 399 p.

Thurman, P., R. McBride, G.D. Dennis, III, and K J. Sulak. 2003. Age and reproduction in three reef–dwelling serranid fishes of the Northeastern Gulf of Mexico Outer Continental Shelf: *Pronotogrammus martinicensis, Hemanthias vivanus* & *Serranus phoebe* (with preliminary observations on the Pomacentrid fish, *Chromis enchrysurus*. USGS/OCS/CSC No. 2003–02. Program Report, U.S. Geological Survey Outer Continental Shelf Studies. http://cars.er.usgs. gov/coastaleco/. (viewed on 02/08/2013).

Vecchione, M. 1987. Commercial fishing for Gulf butterfish, *Peprilus burti*, in the Gulf of Mexico. Marine Fisheries Review 49:14–22.

Vukovich, F.M. and B.W. Crissman. 1986. Aspects of warm rings in the Gulf of Mexico. Journal of Geophysical Research 91:2645–2660.

Watson, W. 1996a. Priacanthidae: Catalufas, bigeyes. In: H.G. Moser, ed. The Early Life Stages of Fishes in the California Current region. California Cooperative Oceanic Fisheries Investigations Atlas 33. Allen Press, Inc., Lawrence, KS, USA, p. 900–903.

Watson, W. 1996b. Pomacentridae: Damselfishes. In: H.G. Moser, ed. The Early Life Stages of Fishes in the California Current region. California Cooperative Oceanic Fisheries Investigations Atlas 33. Allen Press, Inc., Lawrence, KS, USA, p. 1054–1063.

Weaver, D.C. and K.J. Sulak. 2000. Trophic subsidies in the twilight zone: Food web structure of deep reef fishes along the Mississippi–Alabama outer continental shelf. In: M. Mckay and J. Nides, eds. OCS Study MMS 2000–030. Proceedings of the Eighteenth Gulf of Mexico Information Transfer Meeting, Kenner, LA, December 1998, U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA, USA, p. 203–208.

Weaver, D.C. K.J. Sulak, W. Smith–Vaniz, and S.W. Ross. 1999. Community structure and trophic relationships of demersal reef fishes of the Mississippi–Alabama outer continental shelf. In: M. McKay and J. Nides, eds. OCS Study MMS 99–0042. Proceedings of the Seventeenth Gulf of Mexico Information Transfer Meeting, Kenner, LA, December 1997, U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA, USA, p. 286–292.

Weaver, D.C., G.D. Dennis III, and K.J. Sulak. 2002. Community structure and trophic ecology of demersal fishes on the Pinnacles Reef tract. OCS Study MMS–2002–034, USGS BSR 2001–0008. U.G. Geological Survey Biological Sciences Report. U.S. Department of the Interior, Minerals Management Service, New Orleans, LA, USA, 85 p. (available at: http://fl.biology.usgs.gov/coastaleco/USGS_Technical_Report_2001–0008.pdf)

Depuration of Macondo (MC-252) Oil Found in Heterotrophic Scleractinian Corals (*Tubastrea coccinea* and *Tubastrea micranthus*) on Offshore Oil/Gas Platforms in the Gulf

Steve R. Kolian
EcoRigs Non-Profit Organization

Scott Porter
Louisiana Universities Marine Consortium

Paul W. Sammarco
Louisiana Universities Marine Consortium

Edwin W. Cake Jr.
Gulf Environmental Associates

DEPURATION OF MACONDO (MC—252) OIL FOUND IN HETEROTROPHIC SCLERACTINIAN CORALS (*TUBASTREA COCCINEA* AND *TUBASTREA MICRANTHUS*) ON OFFSHORE OIL/GAS PLATFORMS IN THE GULF

Steve R. Kolian[1]*, Scott Porter [1,2], Paul W. Sammarco[1,2], and Edwin W. Cake, Jr.[3]

[1]*EcoRigs Non–Profit Organization, 6765 Corporate Blvd., Suite 1207, Baton Rouge, LA 70809 USA; [2]Louisiana Universities Marine Consortium (LUMCON), 8124 Highway 56, Chauvin, LA 70344 USA; [3]Gulf Environmental Associates, 2510 Ridgewood Road, Ocean Springs, MS 39564 USA; *Corresponding author, email: stevekolian@ecorigs.org*

INTRODUCTION

A cluster of offshore platforms (90, 93, and 94) in the Gulf of Mexico (GOM) located within the Grand Isle block (hereafter, called GI 90 block platforms) possess a diverse invertebrate community, including two species of hard corals. Our dive team conducted more than 30 underwater surveys to observe and collect corals from the GI 90 block oil and gas platforms between 10 July 2005 and 21 October 2010 and found only healthy colonies of corals (Sammarco et al. 2009, 2010, 2012).

The BP/*Deepwater Horizon* MC—252 well blowout oc-

Figure 1. *Examples of the colonies of* Tubastrea coccinea *and* T. micranthus *collected from the GI 90 block platforms.* **A.** *One half of this colony of* T. coccinea *was dead, as indicated by the brownish/black color and loss of tissue, when collected from GI 90 block on 8 May 2011.* **B.** T. coccinea *and* T. micranthus *colonies that showed visible signs of contamination.* **C.** *and* **D.** T. coccinea *colonies that showed visible signs of contamination.*

curred on 20 April 2010 and lasted 84 d. It was the second largest marine spill in recorded history (Joye et al. 2011). It leaked 7.0×10^5 m^3 of crude oil from the sea floor into the northern GOM (Crone and Tolstoy 2010), but was reported to have been capped on 15 July 2010. The GI 90 block platforms are located 168 km WSW of BP's MC—252 spill site, and 60 km south of Port Fourchon, LA.

The impetus for this study began on 8 May 2011, when we noted sporadic mortality of colonies of orange–cup corals, *Tubastrea coccinea* (Lesson, 1829), and green sun corals, *Tubastrea micranthus* (Ehrenberg, 1834; Cnidaria, Anthozoa, Scleractinia) among colonies of coral that appeared to be in good health at the GI 90 block platforms. Colonies of *T. coccinea* which appeared to be healthy displayed markedly less tissue regression and were bright orange in color. The morbid polyps were dark brown or black (Figure 1A–D). The corals were collected, transported to the laboratory, and segregated; healthy–appearing corals were placed in quarantine tanks. After 12 h in quarantine tanks, the coral exudate turned the tank water black and the colonies died within 48 h. After observing this unusual behavior, we decided to document subsequent collection events and analyze the discharged oil to determine its source.

MATERIALS AND METHODS

Survey Sites

The GI 90 block platforms (8°32'37.61"N, 90°03' 56.06"W) are located in 64 m of water. Corals were collected on 11 June and 4 August 2011. The upper 20 m of the 64 m tall sub–surface rig support structures were surveyed via SCUBA and coral samples were scraped from the pilings at 12 and 20 m depths. The survey area was ~6,000 m^2. Upon returning to the research vessel, the coral samples were placed in travel containers with seawater (salinity of 30), kept at 21°C, and transported to shore.

Figure 2. Oil slicks observed over a 10 km wide area during a sampling trip to GI 90 block platforms on 14 September 2011, 13 mo after the BP well had been capped. The foamy beige surface oil was similar in appearance to the oil discharge from coral skeletons (see Figure 4C).

We returned to the GI 90 block platforms on 14 September 2011 for additional coral collections; however, the collection trip was cancelled due to the presence of large crude–oil slicks near the GI 90 block (Figure 2).

Aquarium Studies

11 June 2011 Analysis

Corals were examined and the specimens that appeared to be contaminated with oil were excluded from further observation. About 40 colonies of *T. coccinea* and 10 colonies of *T. micranthus* were placed in the quarantine tanks. The stocking density was about one coral colony per 35 L of seawater and the colonies were maintained in the quarantine tanks for 8 wk. The tank water was changed with GOM seawater (salinity 30) 3 times the first day, twice daily for the following week, and then once daily for the next 4 wk. Water changes were reduced to once weekly from weeks 5 through 8.

4 August 2011 Analysis

Colonies of coral were examined and any that appeared to be contaminated (possessing dead black or brown polyps) were excluded from further observation. About 20 colonies of *T. coccinea* were placed in 57 L quarantine tanks. The stocking density was about one colony per 35 L of seawater. Routine water changes of $\geq$50% of the tank's volume were performed after 2, 8, 16, and 48 h and on day 5. The salinity of the tank water was maintained at 30.

After day 5, the water within the aquaria was filtered using mechanical filter floss and a 0.25 x 0.25 m hydrocarbon adsorbent cloth (Dynamic Adsorbents®). The filter was wrapped in aluminum foil, sealed in a plastic bag, and stored at −20°C to preserve the sample for future analysis.

A sample of the filtered tank water was sent to an analytical chemistry laboratory specializing in crude–oil and related substances (ALS Environmental, Edmonton, Canada) for determination of biomarker profiles. Biomarkers are hydrocarbon molecules derived from formerly living organisms and are present in crude oils at low concentrations (<100 ppm; Wang et al. 2007). The sample biomarker chromotagrams were matched with biomarkers from a BP MC–252 pipeline riser sample. Environmental applications of biomarker fingerprinting have been extensively reviewed elsewhere (Peters and Moldowan 1993; Peters et al. 2005a, 2005b; Wang et al. 2006). The methodology is described in Wang et al. (2007) and Hansen et al. (2007), and has been revised by the European Committee for Standardization (ECS) (2012). The sample was not analyzed for the presence of dispersants.

On day 7, all surviving corals were sacrificed by placing them in fresh water. Three of the colonies were removed from the saltwater quarantine tanks and placed in a 113 L container of fresh water for 28 d. The purpose of the freshwater soaking was to determine if hydrocarbon substances would continue to exude from the deteriorating coral tissues. All photographs were recorded using Sony FX High–Definition video camera and an 18 megapixel Cannon Rebel digital camera.

RESULTS

11 June 2011

Of the 125 coral colonies collected from the platform pilings, 60% showed visible signs of polyp mortality and were excluded from further observation. When the healthy colonies were placed in the quarantine tanks, they released a cloudy white material for 14 h. Several hours later, the aquarium water turned brown and then black. After water changes, the water would change from light brown to black as the corals depurated oil. The corals appeared to purge most of the material within 7 d, but continued to display retracted polyps, exuded mucus, and showed signs of tissue loss for another 5 d. By the end of week 2, the water in
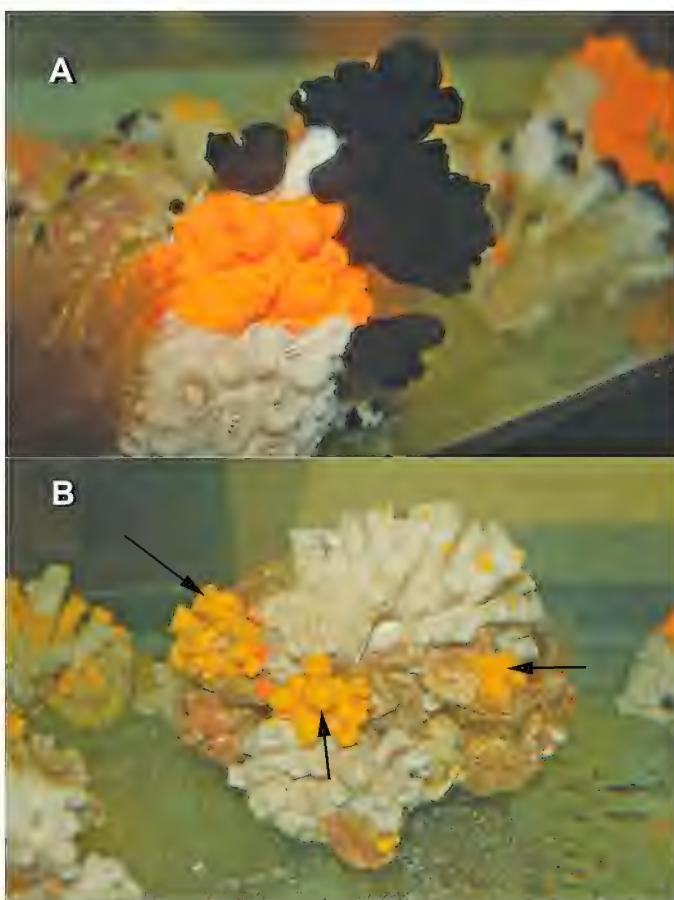
Figure 3. A. Coral polyps that survived 4 wk in quarantine: orange colony = Tubastrea coccinea; dark green colony = Tubastrea micranthus. **B.** "Budding" yellow polyps (arrows), evidence of asexual reproduction, found growing on 10% of the colonies by week 6. Previously, during weeks 1 and 2, the corals had sloughed off muscle tissue from their stomach and calyx, and the skeletons were bleached.

the tanks remained clear overnight and the surviving corals began to extend their tentacles and started feeding on shredded shrimp.

By day 14, 80% of the coral polyps sloughed off injured tissue and transformed into bleached skeletons. The remaining 20% maintained their orange tissue (*T. coccinea*) or dark green tissue (*T. micranthus*) and regained their vigor. After 4 wk in the quarantine tanks, the surviving corals were feeding regularly and exhibiting normal color and behavior (Figure 3A).

By week 6, evidence of asexual reproduction could be seen in *T. coccinea* on the coral skeletons. The yellow coral polyps (Figure 3B) exhibited "budding" or asexual reproduction on the skeletons of recently morbid coral.

4 August 2011

Of the 80 coral colonies transported to the laboratory, 70% showed visible signs of polyp mortality. The remaining 30% that appeared healthy were placed in quarantine tanks. The water turned black after 9 h in the aquaria and corals continued to discharge oil after each water change for 7 d. The black oily water in quarantine tanks (Figure 4A) was

present 48 h after the corals were placed in the aquaria. The foamy surface oil (Figure 4B) continued to appear after 5 d of water changes.

After 7 d in quarantine, 90% of the corals showed signs of substantial tissue regression (>50%), and the cultivation was then terminated for study purposes. Several of the largest colonies of *T. coccinea* were placed in fresh water for 28 d where they continued to discharge crude oil and slough off tissue postmortem (Figure 4C).

Chemical analysis of the hydrocarbon exudate from the corals matched crude oil signatures characteristic of oil from the MC–252 field. ALS Laboratories performed a critical–difference analysis of the calculated ratios of biomarker which revealed that 7 out of 8 ratios obtained from the coral samples matched those obtained for MC–252 crude oil. This suggests, from a weight–of–evidence approach, that the crude oil found in the water of the coral quarantine tanks was a positive match with the crude oil from the MC–252 well.

Discussion

Coral samples have been collected at the GI 90 block platforms since 10 July 2005, including eight occasions between 14 June and 21 October 2010, during and shortly after the MC–252 blowout period. Prior to 8 May 2011, we
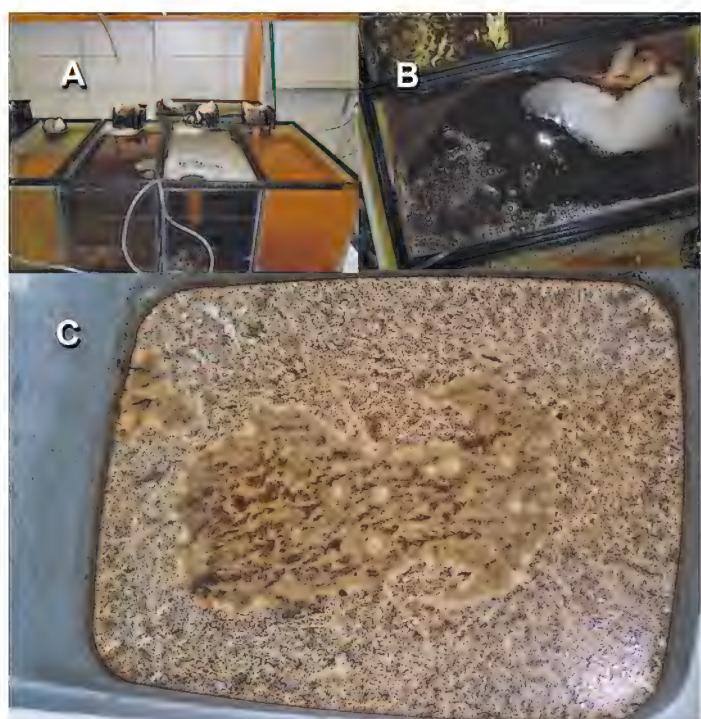
Figure 4. Examples of quarantine aquaria. **A.** The oil ingested by the corals turned the water black after 48 h (10 August 2011). **B.** Day 5 in quarantine tanks (13 August 2011), the corals continued to purge oil. Brownish black foam appeared on the surface. **C.** The dead coral continued to leach beige foamy crude oil after 28 d in fresh water (8 September 2011). The surface oil in the container is similar to crude oil slicks observed offshore (Figure 2).

found no visual evidence of oil discharge in tank water or high polyp mortality among corals during the previous 7 yr of coral collection for research, including during and after the BP oil–well blowout. In the past, we noted that colonies of *T. coccinea* occasionally discharged orange materials during and shortly after transport from the platform to the laboratory, likely from the coelom or muscle tissue. *Tubastrea* spp. are known to respond to stress by producing tubastrine, a mucus material that protects the colony from biological and viral threats (Meyer et al. 2009). Prior to 8 May 2011, the corals in quarantine tanks occasionally discharged orange materials, requiring one water change to clear from the tanks; however, the oil discharges in June and August of 2011 were greater and lasted longer.

Coral behaviors during the June and August 2011 observation periods were similar in that the coral discharged oil for 7 d; however, the first group of corals discharged a white milky material for the first 14 h whereas the second group of corals initially discharged oil that turned the water in the tank black within 9 h. The second observation period was shorter due to the high rate of mortality (90%) of coral polyps. During the 4 August 2011 quarantine period, concentrations of oil were noticeably different from tank to tank; 2 of the 57 L tanks were substantially darker than other tanks.

In addition to scleractinian corals, gorgonians, sponges, tunicates, barnacles, bryozoans, hydrozoans, and oysters inhabit the offshore platforms (Gallaway et al. 1981, Lewbel et al. 1987, Dokken et al. 2000, Carney 2005, Rouse 2009). White et al. (2011) investigated deepwater coral reef habitats (1,370 m) and found crude oil impacts on scleractinian, gorgonian, and antipatharian corals 11 km SW of BP's MC–252 well 3 mo after its reported capping. They describe the appearance of a "brown flocculent material" on the surface of the invertebrates which resemble the materials on the corals found here (see Figure 1B). The coral samples (n = 5) were subjected to chemical analysis and biomarkers were found to match MC–252 oil (White et al. 2012).

The heterotrophic coral on the GI 90 block platforms appear to be assimilating subsurface crude oil. About 3,500 major structures are scattered across the northern GOM (BOEM 2012) and we suggest that some heterotrophic and filter–feeding organisms on the pilings of offshore platforms can be used to determine the source (Wang et al. 2006; Hansen et al. 2007) as well as the horizontal and vertical distributions of BP's MC–252 oil spill. Therefore, by collecting corals from platforms in other areas of the GOM and analyzing the organisms for hydrocarbons, one might be able to determine the geographic range of the MC–252 crude oil plume. The combination of these data could provide geographic insight into the BP MC–252 spill and assist in determining recent and historical exposure to subsurface crude oil.

LITERATURE CITED

Bureau of Ocean Energy Management (BOEM) 2012. Platform database of all structures in the Gulf of Mexico. http://www.gomr.boemre.gov/homepg/pubinfo/freeasci/platform/freeplat.html (viewed on 11/14/12).

Carney, R.S. 2005. Characterization of Algal–Invertebrate Mats at Offshore Platforms and the Assessment of Methods for Artificial Substrate Studies. OCS Study MMS 2005–038. Final Report. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS, New Orleans, LA, USA, 93 p.

Crone, T.J. and M. Tolstoy. 2010. Magnitude of the 2010 Gulf of Mexico oil leak. Science 330:634.

Dokken, Q.R., K. Withers, S. Childs, and T. Riggs, 2000. Characterization and comparison of platform reef communities off the Texas coast. TAMU–CC–0007–CCS. Final Report, Center for Coastal Studies Texas A&M University–Corpus Christi, Texas, USA, 75 p.

European Committee for Standardization (ECS). 2012. Oil spill identification – waterborne petroleum and petroleum products – Part 2: Analytical methodology and interpretation of results based on GC–FID and GC–MS low resolution analyses.

Beuth Verlag, Berlin, Germany, 56 p.

Gallaway, B.J., L.R. Martin, R.L. Howard, G.S. Boland, and G.D. Dennis. 1981. Effects on artificial reef and demersal fish and macrocrustacean communities. The Buccaneer gas and oil field study. Marine Science 14:237–299.

Hansen, A.B., P.S. Daling, L. Faksness, P. Kienhuis, and R. Duss. 2007. Emerging CEN methodology for oil spill identification. Oil spill environmental forensics. In: Z. Wang and S.A. Stout, eds. Oil Spill Environmental Forensics: Fingerprinting and Source Identification, Academic Press, New York, NY, USA, p. 229–256.

Joye, S.B., I.R. MacDonald, I. Leifer, and V. Asper. 2011. Magnitude and oxidation potential of hydrocarbon gases released from the BP oil well blowout. Nature Geoscience 2011:160–164.

Lewbel, G.S., R.L. Randall, and B.J. Gallaway. 1987. Zonation of dominant fouling organisms on northern Gulf of Mexico petroleum platform. Marine Environmental Research 21:199–224.

Meyer, M., F. Delberghe, F. Liron, M. Guillaume, A. Valentin, and M. Guyot. 2009. An antiplasmodial new indole alkaloid from the hard coral *Tubastrea* sp. Natural Product Research 23:178–182.

Peters, K.E. and J.W. Moldowan. 1993. The Biomarker Guide: Interpreting Molecular Fossils in Petroleum and Ancient Sediments. Prentice Hall, Englewood Cliffs, NJ, USA, 363 p.

Peters, K.E., C.C. Walters, and J.W. Moldowan, 2005a. The Biomarker Guide. Volume 1: Biomarkers and Isotopes in the Environment and Human History. Cambridge University Press, Cambridge, UK, 492 p.

Peters, K.E., C.C. Walters, and J.W. Moldowan, 2005b. The Biomarker Guide. Volume 2: Biomarkers and Isotopes in Petroleum Systems and Earth History. Cambridge University Press, Cambridge, UK, 704 p.

Rouse, L. 2009. Evaluation of oil and gas platforms on the Louisiana continental shelf for organisms with biotechnology potential. OCS Study MMS 2009–059. Technical Report, US Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA, xi + 53 p.

Sammarco, P.W., A.D. Atchison, D.A. Brazeau, G.S. Boland, and A. Lirette. 2009. Coral distribution, abundance, and genetics in the Northern Gulf of Mexico: Role of the Flower Garden Banks and Oil/Gas Platforms. Proceedings, US Department of Interior, Minerals Management Service Information Transfer Meeting, New Orleans, LA. USA, 29 April 2009, 23 p.

Sammarco, P.W., S.A. Porter, and S.D. Cairns. 2010. A new coral species introduced into the Atlantic Ocean — *Tubastrea micranthus* (Ehrenberg 1834) (Cnidaria, Anthozoa, Scleractinia): An invasive threat? Aquatic Invasions 2:131–140.

Sammarco, P.W., D.A. Brazeau, and J. Sinclair. 2012. Genetic connectivity in scleractinian corals across the northern Gulf of Mexico: oil/gas platforms, and relationship to the Flower Garden Banks. PLoS ONE 7: e30144.

Wang, Z., S.A. Stout, and M. Fingas. 2006. Forensic fingerprinting of biomarkers for oil spill characterization and source identification. Environmental Forensics 7:105–146.

Wang, Z., C. Yang, M. Fingas, B. Hollebone, U.H. Yim, and J.R. Oh. 2007. Petroleum biomarker fingerprinting for oil spill characterization and source identification. In: Z. Wang and S.A. Stout, eds. Oil Spill Environmental Forensics: Fingerprinting and Source Identification. Elsevier, Inc., Amsterdam, The Netherlands, p. 73–146.

White, H.K., P.Y. Hsing, W. Cho, T.M. Shank, E.E. Cordes, A.M. Quattrini, R.K. Nelson, R. Camilli, A.W. Demopoulos, C.R. German, J.M. Brooks, H. Roberts, W. Shedd, C.M. Reddy, and C.R. Fisher. 2012. Impact of the Deepwater Horizon oil spill on a deep–water coral community in the Gulf of Mexico. Proceedings National Academy of Sciences 109: 20303–20308.

Effects of Closure of the Mississippi River Gulf Outlet on Saltwater Intrusion and Bottom Water Hypoxia in Lake Pontchartrain

Michael A. Poirrier
University of New Orleans, mpoirrie@uno.edu

EFFECTS OF CLOSURE OF THE MISSISSIPPI RIVER GULF OUTLET ON SALTWATER INTRUSION AND BOTTOM WATER HYPOXIA IN LAKE PONTCHARTRAIN

Michael A. Poirrier

Department of Biological Sciences, University of New Orleans, Lakefront Campus, 2000 Lakeshore Drive, New Orleans, LA 70148 USA; email: mpoirrie@uno.edu

INTRODUCTION

Saltwater intrusion from the Mississippi River Gulf Outlet (MRGO) through the Inner Harbor Navigation Canal (IHNC) produced salinity stratification in Lake Pontchartrain which caused episodes of bottom water hypoxia and anoxia from 1968–2009 (Poirrier et al. 2009). This resulted in significant changes in benthic invertebrate community structure which were more distinct and persistent in a 250 km² (100 mi²) area north of the IHNC (Abadie and Poirrier 2001a, 2001b) where more saline water entered.

The MRGO was a deep draft shipping channel which provided a more direct route from the Gulf of Mexico (GOM) to New Orleans. Construction began in 1958 and the MRGO was completed and fully operational in 1968. However, planned economic development along the outlet never materialized and maintenance costs greatly exceeded economic benefits. The MRGO also impacted adjacent wetlands and contributed to the storm surge from Hurricane Katrina that flooded New Orleans (Shaffer et al. 2009). The need to improve hurricane protection and prevent additional damage to coastal resources resulted in its de–authorization as a navigation channel in 2008. The MRGO was closed in July 2009 following the construction of a rock dam across the channel near Bayou La Loutre, about 38 km from the intersection with the Gulf Intracoastal Waterway (GIWW). Flow from the IHNC into Lake Pontchartrain was also blocked by a temporary rock cofferdam, which was completed on 31 October 2010. This was to allow construction of a larger cofferdam and a 30 x 5.5 m (95 x 18 ft) sector gate as well as two 15 x 5.5 m (50 x 18 ft) vertical lift gates for hurricane surge protection. The gates were opened to navigation in July 2012. Construction of the IHNC–Lake Borgne surge barrier, which included a sector gate on the GIWW, a vertical lift gate at Bayou Bievenue and a concrete barrier wall across the MRGO began in May 2009 and was completed in June 2011. It reduced tidal exchange through marshes between Lake Borgne and the MRGO. The floodgates at the mouth of the IHNC were opened on 3 August 2012. Details, including maps of the MRGO and other waterways and design

and status of recent U.S. Army Corps of Engineers projects related to the IHNC and MRGO, are available at www.nolaenvironmental.gov as well as other internet sites.

The past occurrence of saltwater intrusion and associated detrimental low dissolved oxygen (DO) from the MRGO is well documented. Poirrier (1978) was first to report that salinity stratification and associated low bottom DO from MRGO saltwater intrusion occurred in Lake Pontchartrain and discussed possible effects on biota. In this paper, it was assumed that significant intrusion started when the MRGO was completed in 1968. Sikora and Sikora (1982) found through various measures of community structure that a stressed benthic community was present in southern Lake Pontchartrain. They attributed this stress mainly to chemical contamination. However, based on studies conducted in 1976–1978, Junot et al. (1983) found that the principal driver of community change was detrimental low DO associated with saltwater intrusion from the IHNC. Poirrier et al. (1984) collaborated with Schurtz and St. Pè (1984) in a more extensive study that included analysis of contaminants and distribution of anoxia and hypoxia which confirmed that low DO was the major source of stress. Further studies of stratified water circulation were conducted by Georgiou and McCorquodale (2000). Abadie and Poirrier (2000) found that the density of large (> 21 mm) *Rangia cuneata* increased after the cessation of shell dredging in 1990. However, they also documented that large clams were absent from about a 250 km² (100 mi²) area north of the IHNC (Abadie and Poirrier 2001a, 2001b). The absence of clams was attributed to low DO. Additional information on the distribution of anoxia and hypoxia and effects on benthos is summarized by Poirrier et al. (2009) and Shaffer et al. (2009). The goal of this study was to determine if MRGO closure by hurricane flood protection projects stopped saltwater intrusion, salinity stratification, and associated low bottom water DO in Lake Pontchartrain.

MATERIALS AND METHODS

The area of study is in Lake Ponchartrain, LA north of
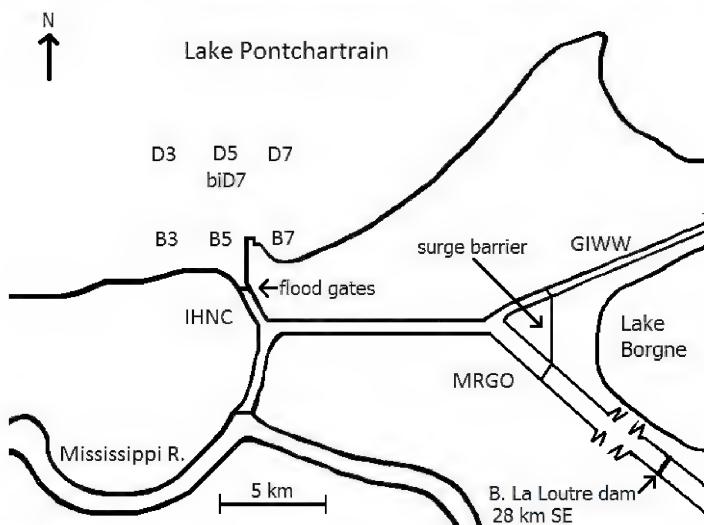
Figure 1. *Map of study area showing the 7 sampling sites in Lake Pontchartrain, and location of the Inner Harbor Navigation Canal (IHNC) and flood gates, the Mississippi River Gulf Outlet (MRGO), the Gulf Intracoastal Waterway (GIWW) and the IHNC–Lake Borgne surge barrier. The MRGO dam at Bayou La Loutre which was off the map scale was included and a gap in the MRGO was used to indicate its true position 28 km SE of the arrow.*

the outlet of the IHNC, an area impacted by changes in water movement through the GIWW, MRGO and IHNC due to flood control structures (Figure 1). Surface salinity and DO (30 cm below surface) and bottom salinity and DO (30 cm from bottom) were measured with either a YSI 85 meter or a YSI 600 sonde. Depth was measured with a calibrated, weighted chain. A benthic invertebrate sampling site biD7 (Poirrier et al. 2009) located 5 km north of the IHNC (30.07500N, 90.04916W) had representative stratification while the MRGO was open. With the exception of fall 2007, it was sampled every fall from 1997 through 2011. Data from this typical site were used to present representative changes that occurred in long–term differences between surface and bottom salinity and dissolved oxygen as flood protection projects that closed the MRGO were completed.

A sampling grid north of the IHNC established in past studies (Poirrier et al. 2009) was used to determine the distribution of stratified water after the 2009 MRGO closure. Surface and bottom temperature, salinity, DO and depth were measured in most months from July to November 2009, March to July 2010 and in August 2012 after the gates at the mouth of the IHNC were opened. Sites near the mouth of the IHNC were sampled first, and sites away from the canal were added until stratification was no longer detected. After closure of the MRGO, it was determined that only 6 sites within 8 km of the IHNC needed to be sampled to confirm the extent of stratification. Sites used with geographic positions and distances from the IHNC were as follows: B3 30.05000N, 90.08333W (4.98 km); B5 30.05000N, 90.05000W (2.44 km); B7 30.05000N, 90.01667W (2.77

km); D3 30.08333N, 90.08333W (7.33 km); D5 30.0833N, 90.05000W (5.90 km); D7 30.08333N, 90.01667W (6.05 km) (Figure 1). Not all sites were sampled on each trip due to bad weather and limited resources. Measurements were not taken during winter months and after storm events that are known to disrupt stratification (Poirrier 1978). Site B7 was not sampled in 2010, because no stratification was detected in 2009. Site D3 was sampled more often upon the elimination of site B7.

RESULTS

A comparison of surface and bottom salinity and DO (Figure 2) from site biD7 indicated that following the closure of the MRGO at Bayou La Loutre in July 2009, salinity differences were less than 0.2 and DO differences were less than 10%, illustrating that the closure stopped significant levels of saltwater intrusion, stratification, bottom water anoxia and hypoxia. Seasonal data from 6 sites near the mouth of the IHNC following the closure of the MRGO supported these results; no differences in surface and bottom salinity values greater than one were present and there was no evidence of hypoxia (DO < 2 mg/l) (Table 1). Differences between surface and bottom salinity ranged from 0.2–0.9. Dissolved oxygen % saturation differences ranged from 0–38 %. Surface and bottom salinity differences greater than 0.1 (the accuracy and resolution of the instrument) occurred on five sampling events; site B5 in July 2009 (0.2), sites B5 and B7 in early August 2009 (0.4 & 0.6), site B5 in late August 2009 (0.9) and site B3 in October 2009. Surface and bottom dissolved
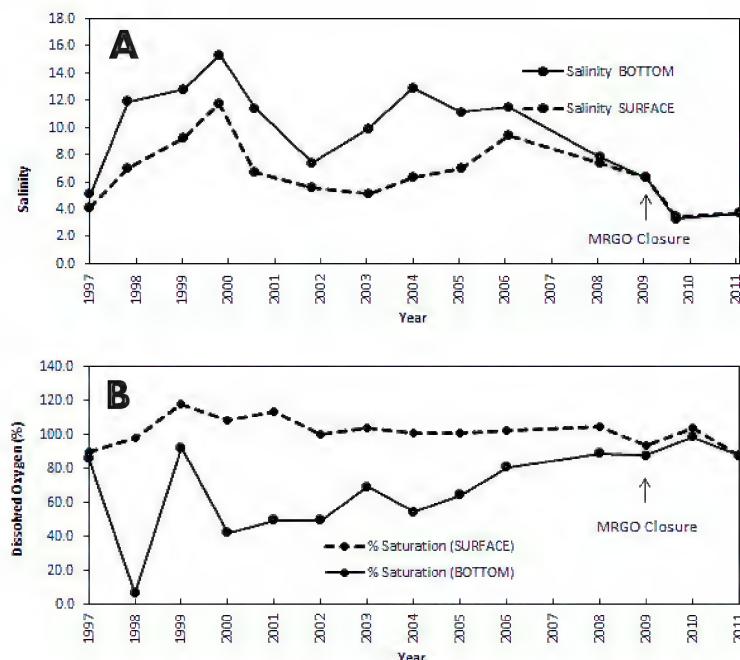

Figure 2. *Comparison of (A) surface and bottom salinity and (B) surface and bottom dissolved oxygen % saturation values at site biD7 located 5 km north of the IHNC before and after the 2009 closure of the MRGO.*

TABLE 1. *Surface and bottom salinity and dissolved oxygen measurements from 6 sites within 8 km of the Inner Harbor Navigation Canal (IHNC) after closure of the Mississippi River Gulf Outlet (MRGO). Numbers in bold represent differences > 0.1 salinity unit or > 10% DO between surface and bottom measurements.*

Date	Site	Surface Salinity	Bottom Salinity	Salinity Differance	Surface DO%	Bottom DO%	%DO Difference	Bottom DO mg/L
7/25/09	B3	5.1	5.2	0.1	99	93	6	7.0
	B5	5.1	5.3	**0.2**	93	89	4	6.4
	B7	5.1	5.1	0	99	95	5	7.0
	D5	4.9	4.9	0	91	91	0	6.7
	D7	4.9	4.9	0	95	89	6	6.6
8/11/09	B3	5.2	5.2	0	97	90	7	6.6
	B5	5.3	5.7	**0.4**	107	69	**38**	5.1
	B7	5.2	5.8	**0.6**	97	82	**15**	6.3
	D5	5.3	5.4	0.1	105	99	6	7.2
8/24/09	B5	6.1	7.0	**0.9**	89	67	**22**	5.0
10/28/09	B3	6.6	6.8	**0.2**	98	93	5	8.1
	B5	6.7	6.8	0.1	102	93	9	8.2
	D5	7.1	7.1	0	93	92	1	8.2
11/19/09	B3	5.9	5.9	0	96	90	6	8.4
	B5	6.2	6.2	0	90	88	2	8.0
	B7	6.4	6.4	0	98	95	3	8.7
3/31/10	B3	1.6	1.6	0	91	88	3	8.4
	B5	1.6	1.6	0	93	89	4	8.5
	D3	1.6	1.7	0.1	94	93	1	8.9
	D5	1.6	1.6	0	94	96	2	9.2
	D7	1.7	1.7	0	97	96	1	9.2
4/29/10	B3	1.9	1.9	0	92	89	3	7.7
	B5	1.9	1.9	0	85	84	1	7.2
	D5	2.0	2.0	0	92	90	2	7.8
5/25/10	B3	1.7	1.7	0	93	90	3	6.9
	B5	1.8	1.8	0	92	91	1	6.9
	D3	1.9	1.9	0	90	86	4	6.7
	D5	1.8	1.8	0	85	86	1	6.6
	D7	1.8	1.8	0	86	86	0	6.6
6/24/10	B3	2.3	2.3	0	95	86	9	6.6
	B5	2.3	2.3	0	84	84	0	6.2
	D3	2.1	2.1	0	94	87	7	6.4
	D5	2.3	2.3	0	92	89	3	6.6
	D7	2.3	2.3	0	93	91	2	6.8
7/10/10	B3	2.4	2.4	0	108	102	6	7.7
	B5	2.4	2.4	0	102	100	2	7.5
	D3	2.4	2.5	0.1	103	100	3	7.5
	D5	2.5	2.5	0	98	95	3	7.1
	D7	2.5	2.5	0	104	102	2	7.6
8/12/12	B3	2.5	2.5	0	101	82	**19**	6.3
	B5	2.6	2.7	0.1	102	92	10	7.0
	D3	2.7	2.7	0	103	91	**12**	6.9
	D5	3.0	3.0	0	105	87	**18**	6.4
	D7	2.6	2.6	0	101	70	**31**	5.3
8/23/12	B3	3.0	3.0	0	102	92	**11**	7.1
	B5	3.0	3.0	0	103	98	5	7.5
	D3	3.2	3.2	0	102	93	9	7.1
	D5	3.0	3.0	0	105	87	**18**	6.4
	D7	2.9	2.9	0	105	93	**12**	7.2

oxygen differences greater than 10% saturation were found on 3 sampling events: 11 August 2009 (B5 [38%; 5.1 mg/l] and B7 [15%; 6.3 mg/l]) and 24 August 2009 (B5 [22%; 5.0 mg/l]). Lower bottom DO, but not hypoxic DO, was associated with stratification. Mean depths for sites throughout the sampling period were: 4.5 ± 0.1 m (B3), 4.9 ± 0.1 m (B5), 3.9 ± 0.1 m (B7), 4.7 ± 0.1 m (D3), 4.7 ± 0.1 m (D5), and

4.8 m ± 0.1 (D7). Low DO values were found at 2 of the deeper sites north of the IHNC; the lowest values were at B5, closest to the IHNC. No evidence of salinity stratification or associated detrimental bottom DO was found in 2010. Data obtained in August 2012 after the IHNC gates at Lake Pontchartrain were opened indicated the continued absence of salinity stratification and hypoxia. However, surface and

bottom DO % saturation differences ranged from 5–31%.

Surface water temperatures ranged from 17.2–30.9°C. Bottom temperatures ranged from 17.2–30.0°C. Water temperature followed typical seasonal trends. Differences between surface and bottom temperatures ranged from 0.0–0.8°C. The temperature differences for the sites with relatively low DO were 0.1°C for B5, 0.2°C for D7 and 0.8°C for D5 on 11 August 2009. Temperature differences within this range were common at other sites that did not have relatively low bottom DO values.

Discussion

Data obtained in this study indicate that MRGO closure at Bayou La Loutre in July 2009 has stopped saltwater intrusion and associated low DO through the IHNC into Lake Pontchartrain. Episodes of extensive anoxia and hypoxia reported in previous studies (Schurtz and St. Pè 1984, Poirrier et al. 2009) were not found after the closure. A trend toward decreased mixing (Figure 2) appears to be present from 2006 to 2008, but this is due to lack of sampling in 2007 and a weather event in 2008. Points on the graph were connected with a line to compare surface and bottom values and do not imply any rate of change over time before the 2009 closure. Some evidence of residual stratification was detected during July, August, and October 2009 (Table 1). No stratification was detected in November 2009 or from March through July 2010. Construction of the temporary IHNC rock cofferdam began in February 2010 and was completed in October 2010. Sampling was discontinued after July 2010 because no stratification was detected and intrusion of higher salinity bottom water appeared to be blocked by the cofferdam which was in the final stages of construction. The IHNC–Lake Borgne surge barrier was under construction as well, and should have stopped any remaining exchange.

The residual stratification observed in 2009 was not surprising. Deep scour channels are present on both sides of the mouth of the INHC. Saline bottom water was probably present in the IHNC and MRGO after closure at Bayou La Loutre. Closure at Bayou La Loutre greatly reduced tidal exchange between the GOM and Lake Pontchartrain through the navigation canals. Any potential exchange was eliminated after placement of the temporary cofferdam for construction of flood protection gates. As in previous studies, temperature differences were not responsible for stratification.

Based on data obtained in August 2012, opening the gates at the IHNC did not produce saltwater intrusion. This is an expected result considering the MRGO was closed and the depth and cross–sectional area of the IHNC at Lake Pontchartrain was reduced by the flood gates. The higher surface to bottom DO differences in this data set may be due to measurements taken only during hot August weather, or to reduced water exchange due to control structures.

Closure of the MRGO has reversed north–to–south salinity gradients in eastern Lake Pontchartrain. Salinity measurement taken prior to closure of the MRGO were higher near the south shore due to saltwater intrusion and decreased with increasing distance from the mouth of the IHNC. Since the closure, salinity is generally higher near the north shore and decreases with decreasing distance to the IHNC.

After the MRGO closure, recovery of the benthic invertebrate community in the 250 km² area north of the IHNC has been slower than anticipated. In addition to the long–term effects of the MRGO, Hurricane Katrina in 2005 caused an acute disturbance that resulted in loss of *R. cuneata* and other community dominants from 815 km² (50% of the bottom) at depths > 3.7 m and limited recovery was observed in 2006 (Poirrier et al. 2008). Fall 2008 benthic invertebrate samples indicated an additional lake–wide disturbance, which further reduced the density of *R. cuneata* and other community–dominant species. This disturbance was more widespread than Katrina and extended to depths < 3.7 m. Ray (2009) studied effects of the 2008 Bonnet Carré Spillway opening on benthos and found similar results. He attributed the stressed community to hurricanes Ike and Gustav. Samples from fall 2009 and 2010 indicate some recovery toward pre–hurricane Katrina conditions, as evidenced by fewer annelids, and more mollusks and arthropods (M. Porrier, unpublished data). However, large *R. cuneata* clams were rare or absent in 2009 and 2010 samples. Recovery studies are still in progress and details will be submitted for publication after ongoing work is completed.

Literature Cited

Abadie, S.W. and M.A. Poirrier. 2000. Increased density of large *Rangia* clams in Lake Pontchartrain after the cessation of shell dredging. Journal of Shellfish Research. 19:481–485.

Abadie, S.W. and M.A. Poirrier. 2001a. Recent trends in water clarity and clam abundance. In: S. Penland, A. Beall, and J. Waters, eds. Environmental Atlas of the Lake Pontchartrain Basin. Lake Pontchartrain Basin Foundation, New Orleans, LA, USA, p.165 .

Abadie, S.W. and M.A. Poirrier. 2001b. *Rangia* Clams as indicators of Hypoxia in Lake Pontchartrain. In: S. Penland, A. Beall, and J. Waters, eds. Environmental Atlas of the Lake Pontchartrain Basin. Lake Pontchartrain Basin Foundation, New Orleans, USA, p. 166.

Georgiou, I.Y. and J.A. McCorquodale. 2000. Salinity stratification from a navigation canal in a shallow lake. In: G. A. Laurence, R. Pieters, and N. Yonemitsu, eds. Stratified flows, Volume 2. International Association for Hydraulic Research, Madrid, Spain, 859–864 p.

Junot, J.A., M.A. Poirrier, and T.M. Soniat. 1983. Effects of saltwater intrusion from the Inner Harbor Navigation Canal on the benthos of Lake Pontchartrain, Louisiana. Gulf Research Reports 3:247–254.

Poirrier, M.A. 1979. Studies of salinity stratification in Southern Lake Pontchartrain near the Inner Harbor Navigation Canal. Proceedings of the Louisiana Academy of Sciences 41:26–35.

Poirrier, M.A., T.A. Soniat, Y.A. King, and L.E. Smith. 1984. An evaluation of the southern Lake Pontchartrain benthos community. Technical report. Louisiana Department of Environmental Quality, Office of Water Resources, Water Pollution Control Division, Baton Rouge, LA, USA, 80 p.

Poirrier, M.A., Z. Rodriguez del Rey, and E.A. Spalding. 2008. Acute disturbance of Lake Pontchartrain benthic communities by Hurricane Katrina. Estuaries and Coasts 31:1221–1228.

Poirrier, M.A., E.A. Spalding, and C.D. Franze. 2009. Lessons learned from a decade of assessment and monitoring studies of submersed aquatic vegetation and benthic invertebrates in Lake Pontchartrain. Journal of Coastal Research. SI 54:88–100.

Ray, G.L. 2009. Response of benthic invertebrate communities following the 2008 Bonnet Carre Spillway Opening. Technical Report. U. S. Army Engineers, New Orleans District, New Orleans, LA, USA, 150 p.

Schurtz, M.H. and K.M. St. Pè. 1984. Water Quality investigations of Environmental conditions in Lake Pontchartrain. Technical Report on Interim findings. Louisiana Department of Environmental Quality, Water Pollution Control Division, Baton Rouge, Louisiana, USA, 85 p.

Shaffer, G. P., J.W. Day, S. Mack, G.P. Kemp, I. van Heerden, M.A. Poirrier, K A. Westphal, D. FitzGerald, A. Melanes, C.A. Morris, R. Bea, and P.S. Penland. 2009. The MRGO Navigation Project: A massive human–induced environmental, economic and storm disaster. Journal of Coastal Research SI 54:206–224.

Sikora W.B. and J.P. Sikora. 1982. Ecological characterization of the benthic community of Lake Pontchartrain, Louisiana. Technical Report. New Orleans District, U.S. Army Corps of Engineers, New Orleans, LA, USA, 214 p.

Distribution and Length Frequency of Invasive Lionfish (*Pterois* sp.) in the Northern Gulf of Mexico

Alexander Q. Fogg
National Marine Fisheries Service

Eric R. Hoffmayer
National Marine Fisheries Service

William B. Driggers III
National Marine Fisheries Service

Matthew D. Campbell
National Marine Fisheries Service

See next page for additional authors

Distribution and Length Frequency of Invasive Lionfish (*Pterois* sp.) in the Northern Gulf of Mexico

Authors

Alexander Q. Fogg, *National Marine Fisheries Service*; Eric R. Hoffmayer, *National Marine Fisheries Service*; William B. Driggers III, *National Marine Fisheries Service*; Matthew D. Campbell, *National Marine Fisheries Service*; Gilmore J. Pellegrin, *National Marine Fisheries Service*; and William Stein, *University of New Orleans*

DISTRIBUTION AND LENGTH FREQUENCY OF INVASIVE LIONFISH (*PTEROIS* SP.) IN THE NORTHERN GULF OF MEXICO

Alexander Q. Fogg[1]*, Eric R. Hoffmayer[1], William B. Driggers III[1], Matthew D. Campbell[1], Gilmore J. Pellegrin[1], and William Stein[2]

[1]*National Marine Fisheries Service, Southeast Fisheries Science Center, Mississippi Laboratories, P.O. Drawer 1207, Pascagoula, MS 39567 USA;* [2]*Nekton Research Laboratory, Pontchartrain Institute for Environmental Sciences, University of New Orleans, 2000 Lakeshore Dr., New Orleans, LA 70148 USA;* *Corresponding author, email: fogg.alex@gmail.com*

INTRODUCTION

In recent years, the prevalence of invasive marine species in United States (US) waters has greatly increased due to anthropogenic factors such as transportation, trade, and aquaculture (Bax et al. 2003). As populations of invasive species increase in a region, they can displace native species, alter community composition and food webs, and change fundamental ecosystem processes (Molnar et al. 2008). Indo—Pacific lionfishes (*Pterois volitans* and *P. miles*) have spread rapidly within US territorial waters of the western North Atlantic Ocean (Morris and Akins 2009, Green et al. 2011). Lionfishes are now known to occur from Massachusetts to the Florida Keys and in the Caribbean Sea and Gulf of Mexico (GOM) (Morris 2009, Schofield 2010). Genetic analysis revealed that the most likely pathway for the introduction of lionfishes into US waters was the result of multiple aquarium releases off the southeast coast of Florida (Betancur—R et al. 2011). The combination of their high spawning frequency (year round, ~every 4 d, Morris 2009) and protracted pelagic larval phase (~26 d, Ahrenholz and Morris 2010), coupled with release in a region with multiple oceanographic currents (e.g., Gulf Stream, Caribbean Current, Yucatan Current and Loop Current) has resulted in the rapid dispersal of lionfishes into the western Atlantic Ocean, including the Caribbean Sea and GOM (Cowen et al. 2006, Betancur—R et al. 2011). The first lionfish was reported from the northern GOM (defined as all US GOM waters within the boundaries of the US Exclusive Economic Zone) in 2006; however, the origin of this specimen has been questioned as it was found dead (Schofield 2009). The next reported sightings occurred during 2010 off southwest Florida, and since that time individuals have been observed as far west as Ewing Bank in continental shelf waters off Louisiana (Schofield 2010).

The purpose of this communication is to document continued spatial expansion of lionfish farther west into the northern GOM. Furthermore, we provide the first length—mass relationships and length frequency information for lionfishes captured within the northern GOM based on data collected as part of a broader on—going study of lionfish life history in this region.

METHODS

Lionfishes were collected opportunistically throughout the northern GOM by spearfishers (divers using pole spear or speargun), commercial trawl operations, and during fishery—independent bottom trawl surveys (National Marine Fisheries Service, Mississippi Laboratories). Collection date, location, and depth associated with capture were provided with each specimen. After collection, most specimens were frozen and subsequently thawed prior to being processed in the laboratory; however, about 20% of specimens were processed in the field shortly after capture. Total length (TL, mm), standard length (SL, mm), and total weight (TW, 0.1 g) were measured. Sex was determined when possible, however, some individuals showed limited development or were damaged during the capture process and could not be sexed. Because lionfishes possess venomous spines, fishers frequently remove the spines and fin rays before fish are landed. Therefore, to describe the relationship between TW and spineless weight (SW), all lionfish landed with spines and fins intact were weighed and then all spines and fin rays were removed with the exception of the caudle fin rays to obtain SW.

Linear regression was performed to describe the relationships between TL and TW, TL and SW, and TL and depth. As weight did not increase linearly with length, those data were log transformed prior to analysis. Analysis of covariance (ANCOVA) was used to determine if there was a significant difference in the relationship of TW and TL with TL as the covariate between sexes. If any variable failed tests of normality and homogeneity of variance, appropriate data transformations were performed. For those specimens whose sex could be identified, a Chi—square test with Yates' correction for continuity was used to determine if the ratio of males to females was different from the expected 1:1. All statistical tests were performed with Statgraphics 5.1 (Statis-
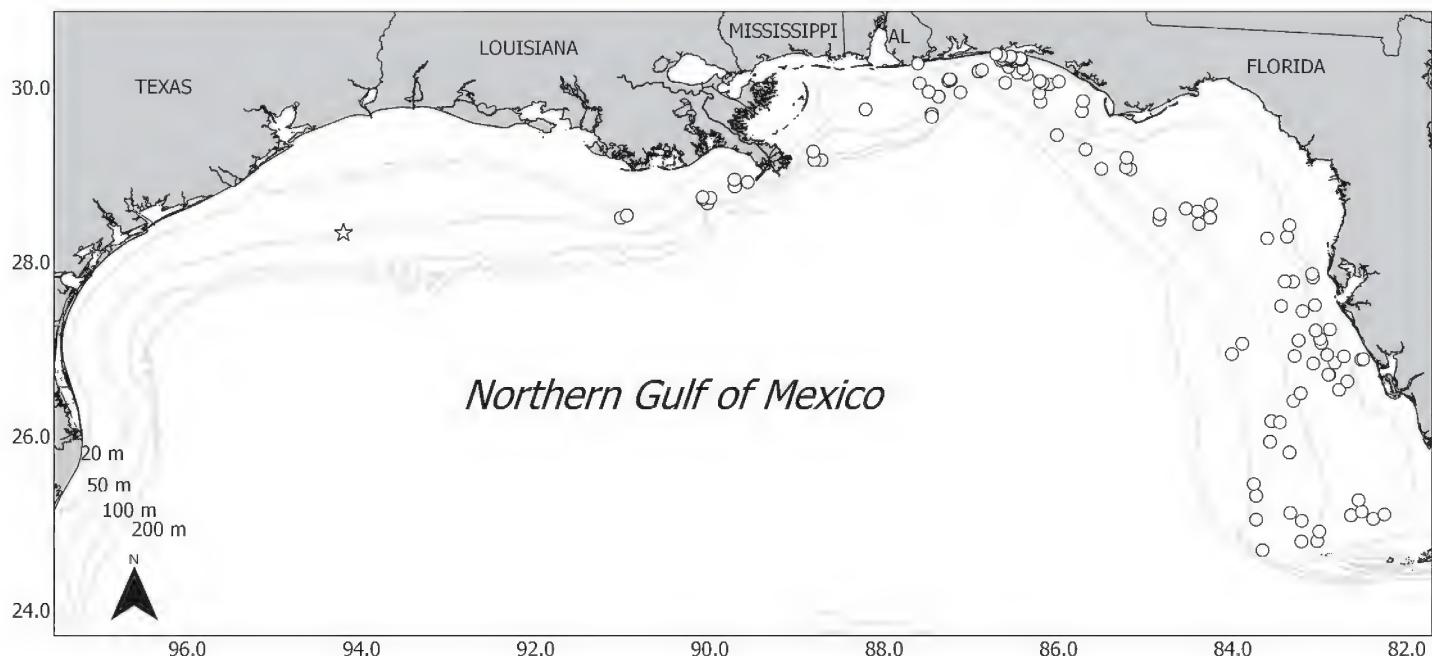
Figure 1. *Locations where lionfish (Pterois spp.) were collected in the northern Gulf of Mexico from March to December 2012. Star indicates the western most collection of Pterois spp. in the northern Gulf of Mexico.*

tical Graphics Corp.) and significance was indicated when $p \leq 0.05$.

RESULTS

From March to December 2012, about 1,500 lionfish were collected from northern GOM waters (Figure 1). A total of 49 lionfish were collected during National Marine Fisheries Service (NMFS) bottom trawl surveys, whereas the rest were collected by spearfishers and commercial trawl operations. Length and weight data were obtained from 582 specimens (Figure 2A). Sex was determined for 294 individuals and the ratio of males to females was 1.03:1 (119 males; 115 females; 60 unknown) and was not significantly

TABLE 1. *Length–mass relationships for* Pterois sp. *specimens collected in the northern Gulf of Mexico. TL = total length (mm), SL = standard length (mm), TW = total weight (g), and SW = spineless weight (g).*

Conversion	n	Equation	r^2
Pooled			
SL to TL	582	SL = 0.7866(TL) −7.5978	0.99
TW to TL	582	log TW = 3.4349(log TL) −5.8608	0.99
SW to TL	582	log SW = 3.4761(log TL) −5.9783	0.99
SW to TW	582	SW = 0.9581(TW) −0.6100	0.99
Male			
TW to TL	119	log TW = 3.3100(log TL) −5.5693	0.97
SW to TL	119	log SW = 3.3265(log TL) −5.6289	0.97
Female			
TW to TL	115	log TW = 3.1437(log TL) −5.1692	0.94
SW to TL	115	log SW = 3.1546(log TL) −5.2160	0.93

different from 1:1 (X^2 = 0.068, p = 0.794). Sex was difficult to assign in fish < 165 mm TL and resulted in 60 specimens with undetermined sex, which ranged in size from 80–163 mm TL. Males and females ranged in length from 126–385 and 157–337 mm TL, respectively (Figure 2B). In July 2012, 8 lionfish were collected by spearfishers about100 km due south of High Island, Texas (28.329°N, −94.1521°W) in 22.9 m of water, which represents the first confirmed lionfish captured in the western GOM. The fish ranged in size from 143–274 mm TL and 31–290 g. The samples consisted of 5 males, one female, and 2 of unknown sex.

Relationships among length and weight are reported in Table 1. There was no significant difference between length–weight relationships by sex (ANCOVA: slope: F = 3.21, p = 0.07; overall: F = 0.40, p = 0.530). There was a significant positive relationship between TW and SW (n = 510, $F_{1,508}$ = 363,315, r^2 = 0.999, p < 0.001), with SW representing about 95% of TW. Specimens were collected at depths ranging from 2.0–84.1 m (mean = 31.8, se = 0.5); however, there was no relationship between TL and collection depth (sexes combined: n = 460, $F_{1,458}$= 0.004, p = 0.951; females: n = 112, $F_{1,110}$ = 0.663, p = 0.417; males: n = 116, $F_{1,114}$= 0.030, p = 0.862).

DISCUSSION

With the exception of a single dead specimen collected off Treasure Island, Florida in 2006, additional movements of lionfish into the GOM were not observed until December 2009 when 2 lionfish were collected in the southern

GOM off the northern Yucatan peninsula, Mexico (Aguilar–Perera and Tuz–Sulub 2010). During 2010, lionfish were observed in the northern GOM with 3 lionfish observed north–northwest of Key West, FL in July, and 2 lionfish reported off the coast of Louisiana within the western GOM in September (Schofield 2010). Combined with our data, lionfish distribution in the northern GOM now extends further west into Texas waters. These observations suggest that if lionfish are not already distributed throughout the continental shelf waters of the entire GOM then they likely will be so in the near future.

The largest lionfish collected in this study was 385 mm TL, and based on a lionfish age–at–length equation developed from fishes collected off North Carolina (Barbour et al. 2011), the back transformed age would be 4.5 y. If northern GOM lionfish growth rates are similar to those off the US east coast, then this specimen was introduced into the northern GOM sometime in early 2008, 2 years prior to the first reported sighting of a live lionfish in this region (Schofield 2009). With larval transport thought to be the primary mechanism for dispersal into the northern GOM (Vasquez–Yeomans et al. 2011), an earlier introduction time of lionfish into this region seems likely. As larger specimens are collected and additional age information becomes available, better estimates of their potential time of introduction into the GOM region will be available.

Based on published studies, the largest lionfish have been collected on the US east coast of North Carolina (Whitfield et al. 2006, Barbour et al. 2011, Muñoz et al. 2011), South Carolina (Meister et al. 2005), north Florida (Ruiz–Carus et al. 2006), and the Bahamas (Morris and Akins 2009, Green et al. 2011), followed by the southern GOM (Aguilar–Perera et al. 2012 and the Caribbean Sea (González et al. 2009, Lasso–Alcalá and Posada 2010). The largest specimens collected from the northern GOM were smaller than lionfish collected from the US east coast, similar in size to specimens from the southern GOM (Aguilar–Perera et al. 2013) and larger than specimens collected from the Caribbean Sea.

While there was no significant effect of depth on sex and size distribution of lionfish collected during this study, this could have resulted from a sampling bias as smaller specimens could have been less apparent on reefs, more difficult to spear, and/or perhaps less 'favored' by the spearfishers. The non–significant effect of depth could also be due to the lack of samples from shallow depths. For example, few samples in our study were collected at depths < 20 m; however, 2 studies have shown smaller lionfish occur in shallower areas when compared to deep reef habitats (Barbour et al. 2010, Claydon et al. 2011). Recent reports from divers in Choctawhatchee and Pensacola Bays along Florida's panhandle, and a recent collection of 2 specimens from shallow waters (~3 m) in coastal Alabama (A. Fogg, unpublished data) demonstrate that lionfish also occur in nearshore wa-
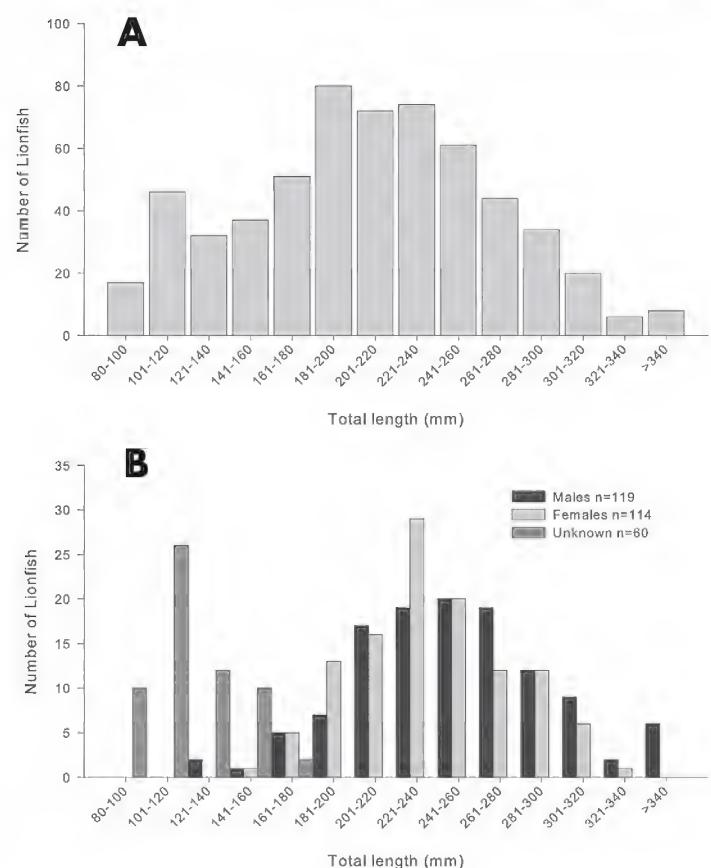
Figure 2. Length frequency of lionfish (Pterois spp.) from the northern Gulf of Mexico. **A.** All measured specimens (n=582). **B.** All specimens examined by sex (n=294)

ters of the northern GOM. This evidence coupled with the findings of Jud et al. (2011) show that it is reasonable that juvenile lionfish can settle in shallow estuarine systems, but a lack of sampling in these areas provides no further evidence besides unconfirmed reports. Furthermore, in the most recent NMFS GOM reef fish video survey, lionfish were observed at Ewing Bank and Pulley Ridge, both of which are located in ~100 m of water (NMFS, unpublished data); however, deepwater habitats are under–represented in diver based studies. Future efforts should focus on collecting lionfishes from all depth ranges equally to address this data gap to determine if there is truly an association among depth, sex and size in the northern GOM.

Since data reported by Schofield (2009, 2010) and in this study were not collected systematically, it has not been possible to establish an accurate rate of expansion for the northern GOM; however, catches of lionfish during fishery–independent surveys suggest the movement of these invasive fishes into the northern GOM has occurred relatively rapidly. During annual groundfish surveys, NMFS conducted about 20,000 bottom trawls in the northern GOM from 1972–2012. During this period no lionfish were collected until 2010 and 2011, when one and 3 lionfish were cap-

tured, respectively. Subsequently, a 16–fold increase in the number of lionfish was observed in 2012 when 49 specimens were captured. Similarly, from 1993–2012, over 7,000 camera array deployments were conducted during NMFS reef fish video surveys throughout the northern GOM from the Dry Tortugas to Brownsville, Texas. No lionfish were observed until 2012, when numerous individuals were recorded on video (exact number not available at the date of submission; NMFS, unpublished data). Importantly, these 2 surveys were conducted on vastly different habitats (e.g., mud to hard bottom vs. reef) so the patchy distribution of reef habitat in the northwestern GOM is unlikely to preclude further expansion into this region.

Based on the rapid expansion of lionfish into the GOM,

combined with the scarcity of predators in the region (Mumby et al. 2011), it is likely sightings and captures of these invasive fishes will increase. To better understand the ecological impacts of lionfish in the GOM, it will be necessary to gain a basic understanding of their biology and trends in population abundance similar to what has been done in other regions of the western North Atlantic Ocean (Whitfield et al. 2006, Morris 2009, Morris and Akin 2009, Morris et al. 2011). Future research should focus on mechanisms of dispersal for lionfish in the northern GOM, not only to understand their population dynamics, but also to understand mechanisms of dispersal for other potentially invasive organisms.

Literature Cited

Aguilar–Perera, A. and A. Tuz–Sulub. 2010. Non–native, invasive red lionfish (*Pterois volitans* [Linnaeus, 1758]: Scorpaenidae), is first recorded in the southern Gulf of Mexico, off the northern Yucatan Peninsula, Mexico. Aquatic Invasions 5:S9–S12.

Aguilar–Perera, A., A. Tuz–Sulub, L. Perera–Chan, M.J. Lopez–Gomez, X.G. Triste, and E.C. Flota. 2012. Lionfish invasion off the northern coast of the Yucatan Peninsula, Mexico, Southern Gulf of Mexico: What do we know? Proceedings of the Gulf and Caribbean Fisheries Institute 64:34–38.

Aguilar–Perera, A., L. Perrera–Chan, and L. Quijano–Puerto. 2013. Record body size for the non–native lionfish *Pterois volitans* (Scorpaeniformes) in the southern Gulf of Mexico. Gulf and Caribbean Research 25:121-123.

Ahrenholz, D.W. and J.A. Morris. 2010. Larval duration of the lionfish, *Pterois volitans* along the Bahamian Archipelago. Environmental Biology of Fishes 88:305–309.

Barbour, A.B., M.L. Montgomery, A.A. Adamson, E. Díaz–Ferguson, and B.R. Silliman. 2010. Mangrove use by the invasive lionfish *Pterois volitans*. Marine Ecology Progress Series 401:291–294.

Barbour A.B., M.S. Allen, T.K. Frazer, and K.D. Sherman. 2011. Evaluating the potential efficacy of invasive lionfish (*Pterois volitans*) removals. PLoS ONE 6:e19666.

Bax, N., A. Williamson, M. Aguero, E. Gonzalez, and W. Geeves. 2003. Marine invasive alien species: a threat to global biodiversity. Marine Policy 27:313–323.

Betancur–R, R., A. Hines, A. Acero Pizarro, G. Ortí, A.E. Wilbur, and D.W. Freshwater. 2011. Reconstructing the lionfish invasion: insights into Greater Caribbean biogeography. Journal of Biogeography 38:1281–1293.

Claydon, J.A.B., J. Batchasingh, M.C. Calosso, S.E. Jacob, and K. Lockhart. 2010. Invasive red lionfish in shallow habitats of the Turks & Caicos Islands. Proceedings of the Gulf and Caribbean Fisheries Institute 63:315–319.

Cowen, R.K., C.B. Paris, and A. Srinivasan. 2006. Scaling of connectivity in marine populations. Science 311:522–527.

Gonzalez, J., M. Grijalba–Bendeck, A.A. Pizarro, and R. Betancur–R. 2009. The invasive red lionfish, *Pterois volitans* (Linnaeus 1758), in the southwestern Caribbean Sea. Aquatic Invasions 4: 507–510.

Green, S.J., J.L. Akins, and I.M. Côté. 2011. Foraging behavior and prey consumption in the Indo—Pacific lionfish on Bahamian coral reefs. Marine Ecology Progress Series. 433:159—167.

Jud, Z.R., C.A. Layman, J.A. Lee, and D.A. Arrington. 2011. Recent invasion of a Florida (USA) estuarine system by lionfish *Pterois volitans* / P. miles. Aquatic Biology 13:21—26.

Lasso—Alcalá, O.M. and J.M. Posada. 2010. Presence of the invasive red lionfish, *Pterois volitans* (Linnaeus, 1758), on the coast of Venezuela, southeastern Caribbean Sea. Aquatic Invasions 5: S53—S59.

Meister, H.S., D.M. Wyanski, J.K. Loefer, S.W. Ross, A.M. Quattrini, and K.J. Sulak. 2005. Further evidence for the invasion and establishment of *Pterois volitans* (Teleostei: Scorpaenidae) along the Atlantic Coast of the United States. Southeastern Naturalist 4:193—206.

Molnar, J.L., R.L. Gamboa, C. Revenga, and M.D. Spalding. 2008. Assessing the global threat of invasive species to marine biodiversity. Frontiers in Ecology and the Environment 6:485-492.

Morris, J.A. 2009. The Biology and Ecology of the Invasive Indo—Pacific Lionfish. Ph.D. thesis. North Carolina State University, Raleigh, NC, USA, 168 p.

Morris, J.A., Jr. and J.L. Akins. 2009. Feeding ecology of invasive lionfish (*Pterois volitans*) in the Bahamian archipelago. Environmental Biology of Fishes 86:389—398.

Morris, J.A., C.V. Sullivan, and J.J. Govoni. 2011. Oogenesis and spawn formation in the invasive lionfish, *Pterois miles* and *Pterois volitans*. Scientia Marina 75:147—154.

Mumby, P.J., A.R. Harborne, and D.R. Brumbaugh. 2011. Grouper as a natural biocontrol of invasive lionfish. PLoS ONE 6:e21510.

Muñoz, R.C., C.A. Currin, and P.E. Whitfield. 2011. Diet of invasive lionfish on hard bottom reefs of the Southeast USA: insights from stomach contents and stable isotopes. Marine Ecology Progress Series 432:181—193.

Ruiz—Carus, R., R.E. Matheson, Jr., D.E. Roberts, Jr., and P.E. Whitfield. 2006. The western Pacific red lionfish, *Pterois volitans* (Scorpaenidae), in Florida: Evidence for reproduction and parasitism in the first exotic marine fish established in state waters. Biological Conservation 128:384—390.

Schofield, P.J. 2009. Geographic extent and chronology of the invasion of non—native lionfish (*Pterois volitans* [Linnaeus 1758] and *P. miles* [Bennett 1828]) in the Western North Atlantic and Caribbean Sea. Aquatic Invasions 4:473—479.

Schofield, P.J. 2010. Update on geographic spread of invasive lionfishes (*Pterois volitans* [Linnaeus, 1758] and *P. miles* [Bennett, 1828]) in the Western North Atlantic Ocean, Caribbean Sea and Gulf of Mexico. Aquatic Invasions 5:S117-S122.

Vasquez—Yeomans, L., L. Carrillo, S. Morales, E. Malca, J.A. Morris, Jr., T. Schultz, and J.T. Lamkin. 2011. First larval record of *Pterois volitans* (Pisces: Scorpaenidae) collected from the ichthyoplankton in the Atlantic. Biological Invasions 13:2635—2640.

Whitfield, P.E., J.A. Hare, A.W. David, S.L. Harter, R.C. Munoz, and C.M. Addison. 2006. Abundance estimates of the Indo—pacific lionfish *Pterois volitans/miles* complex in the western North Atlantic. Biological Invasions 9:53—64.

Notes on the Biology of Invasive Lionfish (*Pterois* sp.) from the Northcentral Gulf of Mexico

Nancy J. Brown-Peterson
University of Southern Mississippi, nancy.brown-peterson@usm.edu

J. Read Hendon
University of Southern Mississippi

NOTES ON THE BIOLOGY OF INVASIVE LIONFISH (*PTEROIS* SP.) FROM THE NORTHCENTRAL GULF OF MEXICO

Nancy J. Brown—Peterson[1]* and J. Read Hendon[2]
[1]*Department of Coastal Sciences, University of Southern Mississippi, 703 East Beach Dr., Ocean Springs, MS 39564 USA; [2]Center for Fisheries Research and Development, Gulf Coast Research Laboratory, 703 East Beach Dr., Ocean Springs, MS 39564 USA;
*Corresponding author, email: nancy.brown—peterson@usm.edu

INTRODUCTION

The red lionfish (*Pterois volitans*) and devil firefish (*P. miles*) are scorpaenids native to the Indo—Pacific, but in the past 20 years have invaded the waters of the western central Atlantic (Morris and Whitfield 2009), including the wider Caribbean and southern Gulf of Mexico (GOM) (Aguilar—Perera and Tuz—Sulub 2010). Lionfish are predators on small fish and crustaceans (Morris and Adkins 2009) and have no known natural predators, so there is increasing concern about the potential ecosystem impacts of the lionfish on native organisms (Albins and Hixon 2011), particularly in light of their range extension to the northern GOM. Much of the initial knowledge regarding the invasive lionfish stems from research conducted in the south Atlantic where lionfish populations first became established (Donaldson et al. 2011), although the more recent proliferation of the species in the wider Caribbean has resulted in much additional knowledge from this region (Morris 2012).

Populations of invasive lionfish in North Carolina and the Bahamas show asynchronous oocyte development and are indeterminate batch spawners (Morris et al. 2011). Furthermore, actively spawning females were captured during all seasons of the year off North Carolina and the Bahamas (Morris et al. 2009) and have relatively low batch fecundity (Morris 2009). However, their ability to spawn throughout the year, as well as their strategy of producing a gelatinous egg mass has been hypothesized to maximize fertilization success (Morris et al. 2011) and may be key to their invasion success. Morris and Akins (2009) examined the diets of lionfish collected in the Bahamas and documented 41 different species of teleost fishes in gut contents, including many smaller fishes that are common prey items for economically important native species (e.g., groupers and snappers).

The first confirmed sightings of lionfish in the GOM occurred in 2010 (Schofield 2010), where they were reported in the southern GOM off the Yucatan Peninsula, Mexico (Aguilar—Perera and Tuz—Sulub 2010); in 2012 more than 1,500 lionfish were collected from northern GOM locations ranging from Florida to Texas (Fogg et al. 2013). However,

other than visual observations (mostly from recreational divers), there is relatively little information on the biology and ecology of lionfish in the northern GOM. The life history of invasive species can vary as they colonize new habitats and areas (Bøhn et al. 2004); thus, biological parameters described for lionfish along the US southeast coast, the Bahamas, and the Caribbean may differ as the species invades the GOM. Recently, Fogg et al. (2013) provided information on length—weight and sex ratios of northern GOM lionfish. Here, we report biological data from 4 lionfish captured during fishery—independent sampling in nearshore GOM waters off Alabama in May 2012.

MATERIALS AND METHODS

Lionfish were collected during routine groundfish surveys, as part of the Southeast Area Monitoring and Assessment Program (SEAMAP) in May 2012, with a 13 m otter trawl (51 mm stretched mesh) that was towed for 30 min at about 2.5 knots. At each collection station, water quality (temperature (°C), salinity and dissolved oxygen (mg/L)) were recorded from bottom water samples collected in Nisken bottles, and depth at the site (m) was recorded.

All species captured in the trawls were enumerated and identified to species; large trawl samples were subsampled for diversity and abundance for most species, but all penaeid shrimp, red snapper (*Lutjanus campechanus*) and lionfish were sorted from the trawl and processed. Lionfish were measured for total length (TL) and standard length (SL) (mm), weighed (g) and frozen on board the ship. In the laboratory, the fish were thawed, and the gonads and stomachs were removed and fixed in 10% neutral buffered formalin. Specimens were deposited in the Gulf Coast Research Laboratory Museum (accession numbers GCRL 36546 (station E1007) and GCRL 32547 (station E1108)). While *P. volitans* is more common along the southeastern United States and Bahamas than *P. miles*, morphological identification of the species is not accurate in this region (Hamner et al. 2007); genetic identification to species has not been done on these
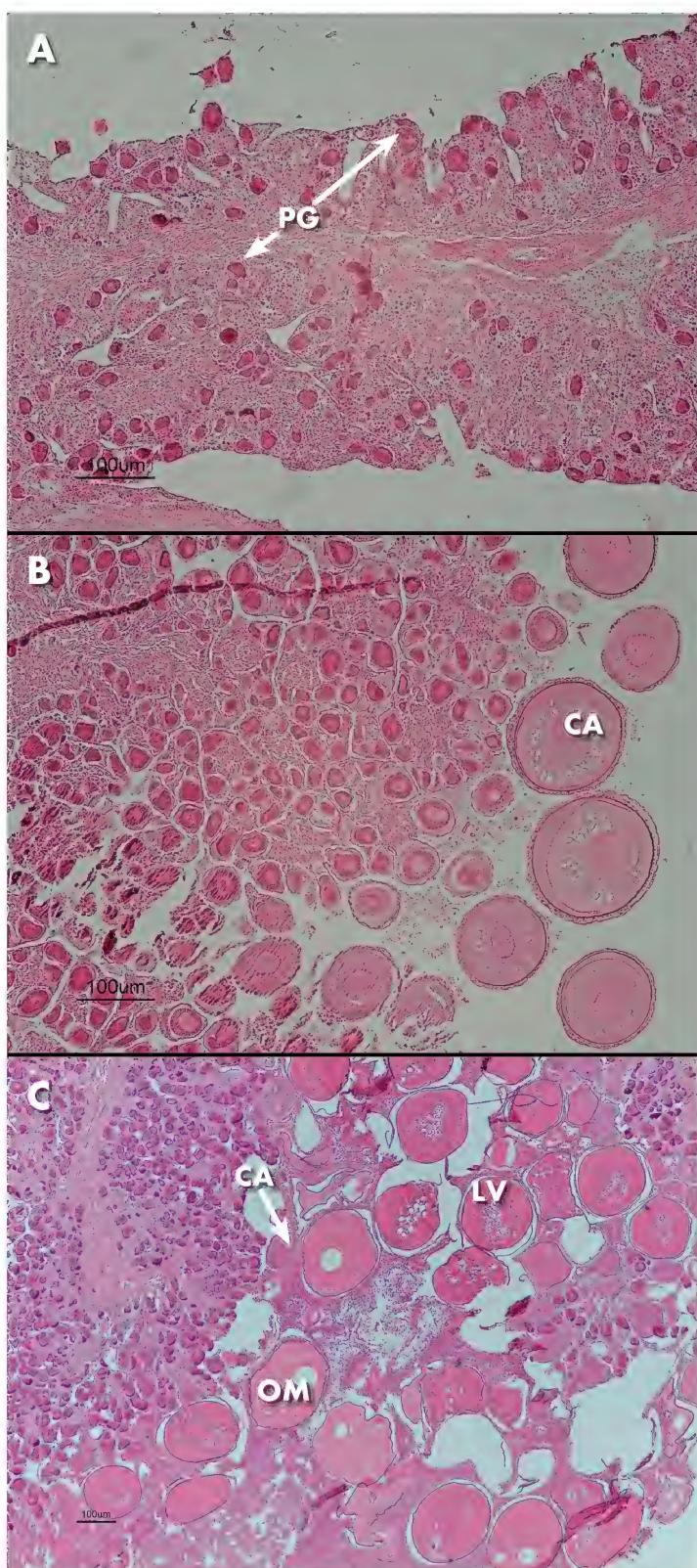
Figure 1. *Photomicrographs of ovarian tissue from lionfish captured on 30 May 2012 from the northcentral Gulf of Mexico. **A.** Immature (149 mm TL). **B.** Developing (188 mm TL). **C.** Spawning capable (213 m TL). CA – cortical alveolar oocyte; LV – late vitellogenic oocyte; OM – oocyte maturation; PG – primary growth oocyte.*

specimens, and therefore they are referred to collectively as lionfish in this paper.

Stomach contents were examined to determine abundance of prey items to the lowest possible taxonomic level. Gonadal tissue was rinsed overnight in running tap water, dehydrated in a series of graded ethanols, cleared in xylene substitute and embedded in paraffin following standard histological techniques. Tissues were sectioned at 4 μm, stained in Hematoxylin 2 and Eosin Y (Richard Allen Scientific), and gonadal maturity phases were assigned following Brown–Peterson et al. (2011).

RESULTS

Lionfish were captured at SEAMAP stations E1108 (29°46.38'N, 88°15.98'W) and E1007 (29°46.52'N, 87°46.83'W) on 30 May 2012 at 1000 and 1500 CDT, respectively. These relatively shallow water (35 m) stations are about 50 km south of the mouth of Mobile Bay, AL. Two lionfish were captured at each station, and ranged in size from 149–213 mm TL (Table 1). Only the gender of the largest fish captured could be determined macroscopically.

TABLE 1. *Biological data for lionfish captured in 30 May 2012 in the northcentral Gulf of Mexico. U = unknown.*

ID	TL (mm)	SL (mm)	Weight (g)	Sex
E1108–1	188	137	88.7	F
E1108–2	149	102	31.8	F
E1007–1	213	161	138.7	F
E1007–2	162	118	52.1	U

Water quality characteristics were similar at both stations (Table 2). Bottom topography of both stations showed some structure; the trawl net tore at station E1007, indicating the presence of hard reef substrate. The species composition of native fishes taken in the same collections (Table 3) also suggests stations E1108 and E1007 had hard bottom substrates.

Stomach content analyses revealed prey material in 3 of the 4 specimens, including fish. However, all prey material had been digested to the point that positive identification of individual items was not possible to the family or species level. Histological analysis revealed that 3 of the 4 lionfish were females; the gender of the fourth fish is unknown due to lack of macroscopic identification of gonadal tissue. The smallest fish captured (149 mm TL) was an immature female, with scattered primary growth (PG) oocytes throughout the ovary (Figure 1A). A larger female (188 mm TL) was in the developing phase, with cortical alveolar (CA) oocytes (Figure 1B), which indicates the beginning of gonadal recrudescence. The largest fish captured (213 mm TL) was a female in the spawning capable phase, exhibiting late vitellogenic oocytes (Vtg3) as well as some oocytes

118

TABLE 2. *Water quality parameters of SEAMAP stations in the northcentral Gulf of Mexico where lionfish were captured on 30 May 2012. Depth of both stations is 35 m.*

Parameter	Station E1108	Station E1007
Bottom Temperature (°C)	21.8	22.9
Bottom Salinity	35.7	35.8
Bottom Dissolved Oxygen (mg/L)	5.9	7.2

just beginning to enter oocyte maturation (OM; Figure 1C). Asynchronous oocyte development is evident in the spawning capable female, indicating this is a batch spawning species. The presence of this fish suggests that lionfish are most likely spawning in the northcentral GOM.

DISCUSSION

Despite the proliferation of lionfish in the GOM since 2010, little biological data has been reported on the species from the region. Recent information on length–weight relationships and sex ratios of northern GOM lionfish (Fogg et al. 2013) provides insights into the growing lionfish population. Spearfishing divers have collected most of the lionfish specimens in the northern GOM from reefs and rigs (A. Fogg, pers. comm., National Marine Fisheries Service, Pascagoula, MS); however, the data we present off Alabama are based on specimens collected during a fishery–independent SEAMAP trawl survey. We collected lionfish over or near hard bottom with relief, as evidenced by both the species composition of other fishes collected in the same trawl (see Table 3) and by the torn net at station E1007 that was undoubtedly damaged by bottom structure.

Invasive lionfish are known to inhabit reefs along the southeast Atlantic coast and in the Caribbean, where their impacts on native fish fauna and the food web are detrimental (Albins and Hixon 2008, 2011, Morris et al. 2009, Arias–Gonzales et al. 2011). They feed mostly on small–bodied teleost reef fish in the Bahamas (Morris and Atkins 2009), and food web models with and without lionfish suggest that lionfish have a detrimental effect on small carnivorous and herbivorous fishes (Arias–Gonzales et al. 2011). Although the stomach contents of the 4 fish we examined were inconclusive since they were mostly digested, clearly lionfish from the northcentral GOM are eating teleost fish. Lionfish show active feeding during the mornings in the Bahamas (0800–1100 h, Morris and Akins 2009). However, partially digested and unidentifiable prey in our fish collected at 1000 and 1500 h may suggest that feeding occurs at different times in the northcentral GOM. Few of the known prey of lionfish from the Bahamas were collected in the trawls in our study; however, lionfish are known to be non–specific piscivores (Morris and Akins 2009), and are thus no doubt eating dominant fishes on the GOM reefs such as sciaenids,

lutjanids and sparids.

Analysis of ovarian tissue suggests lionfish reproduction in the GOM is similar to that seen in North Carolina and the Bahamas (Morris 2009; Morris et al. 2011), with asynchronous oocyte development and the probability of continuous egg production when environmental conditions are favorable. Seasonality of the spawning season in its native range is currently unknown, but collections off North Carolina and the Bahamas suggest that invasive lionfish are able to spawn year–round, with an estimated spawning frequency of every 3–4 days (Morris 2009). Based on data collected in this study, female lionfish in the GOM are sexually mature by 188 mm TL and spawning capable by 213 mm TL; this is similar to findings in North Carolina, where female lionfish > 170 mm TL are sexually mature (Morris 2009).

TABLE 3. *Fish species captured in SEAMAP trawl collections with lionfish from the northcentral Gulf of Mexico on 30 May 2012.*

Parameter	Number at E1108	Number at E1007
Engraulidae		
Anchoa hepsetus	37	
Clupeidae		
Harengula jaguana	6	
Carangidae		
Trachurus lathami	50	
Lutjanidae		
Lutjanus campechanus	13	
Rhomboplites aurorubens		3
Gerreidae		
Eucinostomus argenteus	12	3
Haemulidae		
Haemulon aurolineatum	1	1
Sparidae		
Lagodon rhomboides	62	
Stenotomus caprinus	1054	
Sciaenidae		
Cynoscion arenarius	31	
Cynoscion nothus	6	
Equetus umbrosus		1
Leiostomus xanthurus	175	
Micropogonias undulatus	1004	
Mullidae		
Mullus auratus	6	
Paralichthyidae		
Syacium gunteri		2
Syacium papillosum		1

The data presented here provide preliminary information on the invasive lionfish in the northcentral GOM, and suggests that life history characteristics contributing to their rapid and successful invasion of and establishment in the western North Atlantic and Caribbean regions are similar in the GOM. However, life history traits of invasive fishes can

change during the invasion process as new habitats are colonized (Bøhn et al. 2004, Gibbs et al. 2008, Gutowsky and Fox 2012), underscoring the need for additional, comprehensive biological data on lionfish from the GOM as their abundance increases and they become established.

Acknowledgments

Literature Cited

Aguilar—Perera, A. and A. Tuz—Sulub. 2010. Non—native, invasive red lionfish (*Pterois volitans* [Linnaeus, 1758]: Scorpaenidae), is first recorded in the southern Gulf of Mexico, off the northern Yucatan Peninsula, Mexico. Aquatic Invasions 5:S9—S12.

Albins, M.A. and M.A. Hixon. 2008. Invasive Indo—Pacific lionfish (*Pterois volitans*) reduce recruitment of Atlantic coral—reef fishes. Marine Ecology Progress Series 367:233—238.

Albins, M.A. and M.A. Hixon. 2011. Worst case scenario: potential long—term effects of invasive predatory lionfish (*Pterois volitans*) on Atlantic and Caribbean coral—reef communities. Environmental Biology of Fishes DOI:10.1007/s10641—011—9795—1.

Arias—González, J.E., C. González—Gándara, J.L. Cabrera, and V. Christensen. 2011. Predicted impact of the invasive lionfish *Pterois volitans* on the food web of a Caribbean coral reef. Environmental Research 111:917—925.

Bøhn, T., T.O. Terje Sandlund, P.A. Amundesen, and R. Primicerio. 2004. Rapidly changing life history during invasion. Oikos 106:138—150

Brown—Peterson, N.J., D.M. Wyanski, F. Saborido—Rey, B.J. Macewicz, and S.K. Lowerre—Barbieri. 2011. A standardized terminology for describing reproductive development in fishes. Marine and Coastal Fisheries 3:52—70.

Donaldson, T.J., D. Benavente, and R. Diaz. 2011. Why are lionfishes (*Pterois*, Scorpaenidae) so rare in their native range? Proceedings of the Gulf and Caribbean Fisheries Institute 63:352—359.

Fogg, A.Q., E.R. Hoffmayer, W.B. Driggers III, M.D. Campbell, G.J. Pellegrin, and W. Stein. 2013. Distribution and length frequency of invasive lionfish (*Pterois* sp.) in the northern Gulf of Mexico. Gulf and Caribbean Research 25:111-115.

Gibbs, M.A., G.H. Shields, D.W. Lock, K.M. Talmadge, and T.M. Farrell. 2008. Reproduction of an invasive exotic catfish *Pterygoplichthus disjunctivus* in Volusia Blue Spring, Florida, USA. Journal of Fish Biology 73:1562—1572.

Gutowsky, L.F.G. and M.G. Fox. 2012. Intra—population variability of life—history traits and growth during range expansion of the invasive round goby, *Neogobius melanostomus*. Fisheries Management and Ecology 19:78—88.

Hamner, R.M., D.W. Freshwater, and P.E. Whitfield. 2007. Mitochondrial cytochrome b analysis reveals two invasive lionfish species with strong founder effects in the western Atlantic. Journal of Fish Biology 71 (Suppl. B):214-222.

Johnston, M.W. and S.J. Purkis. 2011. Spatial analysis of the invasion of lionfish in the western Atlantic and Caribbean. Marine Pollution Bulletin 62:1218—1226.

Morris, J.A., Jr. 2009. The biology and ecology of invasive Indo—Pacific lionfish. PhD thesis, North Carolina State University, Raleigh, NC, USA, 168 p.

Morris, J.A., Jr. (Ed.). 2012. Invasive Lionfish: A Guide to Control and Management. Gulf and Caribbean Fisheries Institute Special Publication Series Number 1, Marathon, FL, USA, 113 p.

Morris, J.A. and J.L. Akins. 2009. Feeding ecology of invasive lionfish (*Pterois volitans*) in the Bahamian archipelago. Environmental Biology of Fishes 86:389—398.

Morris, J.A. and P.E. Whitfield. 2009. Biology, ecology, control and management of the invasive Indo—Pacific lionfish: an updated integrated assessment. NOAA Technical Memorandum NOS NCCOS 99:1—57.

Morris, J.A., Jr., J.L. Akins, A. Barse, D. Cerino, D.W. Freshwater, S.J. Green, R.C. Muñoz, C. Paris, and P.E. Whitfield. 2009. Biology and ecology of the invasive lionfishes, *Pterois miles* and *Pterois volitans*. Proceedings of the Gulf and Caribbean Fisheries Institute 61:409—414.

Morris, J.A., Jr., C.V. Sullivan, and J.J. Govoni. 2011. Oogenesis and spawn formation in the invasive lionfish, *Pterois miles* and *Pterois volitans*. Scientia Marina 75:147—154.

Schofield, P. 2010. Update on geographic spread of invasive lionfishes (*Pterois volitans* [Linnaeus, 1758] and *P. miles* [Bennett, 1828]) in the Western North Atlantic Ocean, Caribbean Sea and Gulf of Mexico. Aquatic Invasions 5:S117-S122.

Record Body Size for the Red Lionfish, *Pterois volitans* (Scorpaeniformes), in the Southern Gulf of Mexico

Alfonso Aguilar-Perera
Universidad Autonoma de Yucatan, Mexico

Leidy Perera-Chan
Universidad Autonoma de Yucatan, Mexico

Luis Quijano-Puerto
Universidad Autonoma de Yucatan, Mexico

SHORT COMMUNICATION

RECORD BODY SIZE FOR THE RED LIONFISH, *PTEROIS VOLITANS* (SCORPAENIFORMES), IN THE SOUTHERN GULF OF MEXICO

Alfonso Aguilar–Perera*, Leidy Perera–Chan, and Luis Quijano–Puerto

*Departamento de Biología Marina, Campus de Ciencias Biológicas y Agropecuarias, Universidad Autónoma de Yucatán, Km. 15.5, carretera Mérida–Xmatkuil, A.P. 4–116 Itzimná, C.P. 97315, Mérida, Yucatán, MÉXICO *Corresponding author email: alfaguilar@gmail.com*

INTRODUCTION

Non–native species are those that have been transported, via human actions, from one continent and introduced into another (Lockwood et al. 2007). In the 1980s, red lionfish, *Pterois volitans* (Linnaeus, 1758), traded in the US aquarium industry from the Pacific Ocean, was introduced into the coral reefs off Florida´s coast by aquarium hobbyists (Morris and Whitfield 2009). It is unknown how this introduction occurred, but after more than 20 years the red lionfish population is widespread, occupying a large portion of the Western Atlantic (Schofield 2010) where it represents a threat to the marine ecosystem (Green et al. 2012). The population appears impossible to eradicate since it can live to depths up to 100 m, where individuals consume native small crustaceans and reef fish (Morris and Akins 2009, Green et al. 2012).

Relatively nothing is known regarding biological aspects of the red lionfish in the southern Gulf of Mexico (GOM), despite this being the area of the GOM where lionfish were first detected in late 2009 (Aguilar–Perera and Tuz–Sulub 2010). The present note aims to document the presence of a large–bodied *P. volitans* whose size is the maximum ever recorded in the GOM.

METHODS

During April to July 2010, we hosted workshops for local lobster–diver fishers from the northern coast of the Yucatan Peninsula, Mexico (Aguilar–Perera et al. 2012), and provided them with information on lionfish biology and the chronology of its invasion in the region. These workshops aimed to educate participants on how to collect and document any lionfish they may encounter. On 8 August 2011 at 1400 h, a local diver–fisher speared a large specimen at 33 m in waters off Cayo Arenas (22°07'25"N, 91°23'24"W). Cayo Arenas is a key located on the Campeche Bank 167 km off the northwestern Yucatan Peninsula, Mexico, in the southern GOM. The collected specimen was brought to the laboratory where it was taxonomically identified following Schultz (1986), measured (mm total length, TL) and

weighed (g total weight, TW). Its body cavity was inspected and stomach contents analysed.

RESULTS AND DISCUSSION

The specimen captured in Cayo Arenas was identified as *Pterois volitans* based on meristics (XIII–11 dorsal fin, III–7 anal fin), measured 390 mm TL, and weighed 1,090 g TW (Figure 1). When inspecting its digestive tract and stomach, two fish prey were found: *Haemulon* spp. (47 mm TL) and *Eugerres* spp. (33 mm TL). The lionfish was a female, but its gonads (7 g) were not reproductively active; however, there was presence of mesenteric fat in the body cavity, which may

Figure 1. Red lionfish (Pterois volitans; 390 mm TL) caught off Cayo Arenas in August 2011 by a diver–fisher in the southern Gulf of Mexico.

be indicative of the onset of reproduction.

The specimen caught off the Cayo Arenas exhibited the maximum body size ever recorded for *Pterois volitans* in the GOM. During late 2009, a local diver–fisher captured the first red lionfish off the northern coast of the Yucatan Peninsula, Mexico (Aguilar–Perera and Tuz–Sulub 2010); however, this fish was relatively small (137 mm TL) compared to most lionfish off the eastern US coast and The Bahamas. The large body size record (390 mm TL) now documented in this work endorses the fact that individuals of *P. volitans* have no problems finding food resources in the GOM. Since 2010, local fishers have captured an increasing number of *P. volitans* (n = 445; 90 to 274 mm TL) off the northern Yucatan Peninsula (Aguilar–Perera et al. 2012).

TABLE 1. Maximum body size records for the red lionfish, Pterois volitans, in the western Atlantic and Pacific Oceans, including the new record in the Gulf of Mexico (GOM). TL = Total length.

Country	Locality	TL (mm)	References
Bahamas	Unknown	420	Morris and Akins (2009)
Turks & Caicos	South Caicos	277	Claydon et al. (2009)
USA	North Carolina	450	Whitfield et al. (2007)
	Florida	474	James Morris (Pers. Comm.)
Cuba	Archipielago de Sabana—Camagüey	170	Chevalier et al. (2008)
Mexico	Cayo Arenas (GOM)	390	This study
Honduras	Roatan Marine Park	381	Biggs and Olden (2011)
Bonaire	Unknown	295	Poole and College (2010)
Venezuela	Gran Roque, Los Roques Archipelago National Park	220	Lasso—Alcalá and Posada (2010)
Colombia	Ratón Place, San Andrés Island	235	González et al. (2009)
Pacific Ocean	Unknown	380	Randall et al. (1990)

In the Western Atlantic, *P. volitans* show larger body sizes than those in their native Pacific Ocean (Table 1). Maximum recorded body sizes in the western Atlantic are 474 mm TL (J. Morris, pers. comm., National Marine Fisheries Service, Beaufort, NC) and 450 mm TL (Whitfield et al. 2007), whereas 380 mm TL is the largest reported length from the Pacific (Randall et al. 1990). Fast growth and high abundance are two common traits displayed by invasive species (Morris and Whitfield 2009), and the presence of larger lionfish in the GOM and western Atlantic than in the lionfish's native Pacific suggests that growth, survival and/or longevity may be enhanced in non—native waters. Those life—history characteristics, the species' broad range of prey items (Morris and Akins 2009), and the apparent absence of known predators indicate that *P. volitans* are indeed a threat to the marine ecosystem.

LITERATURE CITED

Aguilar—Perera, A. and A. Tuz—Sulub. 2010. Non—native, invasive red lionfish (*Pterois volitans* [Linnaeus 1758]: Scorpaenidae) is first recorded in the southern Gulf of Mexico, off the northern Yucatan Peninsula, Mexico. Aquatic Invasions 5:S9—S12.

Aguilar—Perera, A., A. Tuz—Sulub, L. Perera—Chan, M.J. López—Gómez, X. González—Triste, and E. Carillo—Flota. 2012. Lionfish invasion off the northern coast of the Yucatan Peninsula, Mexico, Southern Gulf of Mexico: What do we know? Proceedings of the Gulf and Caribbean Fisheries Institute 64:34—38

Biggs, C.R. and J.D. Olden. 2011. Multi—scale habitat occupancy of invasive lionfish (*Pterois volitans*) in coral reef environments of Roatan, Honduras. Aquatic Invasions 6:347—353.

Claydon, J.A.B., M.C. Calosso, and S.E. Jacob. 2009. The red lionfish invasion of south Caicos, Turks & Caicos Islands. Proceedings of the Gulf and Caribbean Fisheries Institute 61:400—402.

Chevalier, P.P., E. Gutiérrez, D. Ibarzabal, S. Romero, V. Isla, J. Calderín, and E. Hernández. 2008. Primer registro de *Pterois volitans* (Pisces: Scorpaenidae) para aguas cubanas. Solenodon 7:37—40.

González, J., M. Grijalba—Bendeck, A. Acero, and R. Betancur—R. 2009. The invasive red lionfish, *Pterois volitans* (Linnaeus 1758), in the southwestern Caribbean Sea. Aquatic Invasions 4:507—510.

Green, S.J., J.L. Akins, A. Maljković, and I.M Côté. 2012. Invasive lionfish drive Atlantic coral reef fish declines. PLoS ONE 7: e32596.

Lasso—Alcalá, O.M. and J.M. Posada J.M. 2010. Presence of the invasive red lionfish, *Pterois volitans* (Linnaeus, 1758), on the coast of Venezuela, southeastern Caribbean Sea. Aquatic Invasions 5:53—59.

Lockwood, J.L., M.F. Hoopes, and M.P. Marchetti. 2007. Invasion Ecology, Blackwell, Oxford, Great Britain, 312 p.

Morris, J.A., Jr. and J.L Akins. 2009. Feeding ecology of invasive lionfish (*Pterois volitans*) in the Bahamian archipelago. Environmental Biology of Fishes 86:389–398.

Morris, J.A., Jr. and P.E. Whitfield. 2009. Biology, ecology, control and management of the invasive Indo–Pacific lionfish: an updated integrated assessment. NOAA Technical Memorandum NOS NCCOS 99:1–57.

Poole, T. and C. College. 2011. The sensitivity of the invasive lionfish, *Pterois volitans*, to parasitism in Bonaire, Dutch Caribbean. Physis Journal of Marine Science 9:44–49.

Randall, J.E., G.R. Allen, and R.C. Steene. 1990. Fishes of the Great Barrier Reef and Coral Sea, University of Hawaii Press, Honolulu, HI, USA, 557p.

Schofield, P.J. 2010. Update on geographic spread of invasive lionfishes (*Pterois volitans* [Linnaeus, 1758] and *P. miles* [Bennett, 1828]) in the Western North Atlantic Ocean, Caribbean Sea and Gulf of Mexico. Aquatic Invasions 5:S117–S122.

Schultz, E.T. 1986. *Pterois volitans* and *Pterois miles*: two valid species. Copeia 1986:686–690.

Whitfield, P.E., J.A Hare, A.W. David, S.L. Harter, R.C. Muñoz, and C.M. Adisson. 2007. Abundance estimates of the Indo–Pacific lionfish *Pterois volitans/miles* complex in the Western North Atlantic. Biological Invasions 9:53–64.

Effects of Black Mangrove (*Avicennia germinans*) Expansion on Saltmarsh (*Spartina alterniflora*) Benthic Communities of the South Texas Coast

Jessica Lunt
Texas A&M University, Corpus Christi

Kimberly McGlaun
Texas Department of Transportation

Elizabeth M. Robinson
Louisiana State University

SHORT COMMUNICATION

EFFECTS OF BLACK MANGROVE (*AVICENNIA GERMINANS*) EXPANSION ON SALTMARSH (*SPARTINA ALTERNIFLORA*) BENTHIC COMMUNITIES OF THE SOUTH TEXAS COAST

Jessica Lunt*, Kimberly McGlaun[1], and Elizabeth M. Robinson[2]

*Department of Life Sciences, Texas A&M University– Corpus Christi, 6300 Ocean Dr., Corpus Christi, TX 78412 USA; [1]Current Address: Texas Department of Transportation, 1701 South Padre Island Dr., Corpus Christi, TX 78416 USA; [2]Current Address: Department of Oceanography and Coastal Science, 1209 Energy, Coast, and Environment Building, Louisiana State University, Baton Rouge, LA 70803 USA; *Corresponding author email: jlunt@tamucc.edu*

INTRODUCTION

Black mangroves are becoming more common in several areas along the Texas coast and encroaching on existing saltmarsh habitats (Everitt et al. 2007, 2008, 2010). Expansion has been documented through the use of aerial photography on South Padre Island (Everitt et al. 2007, 2008, 2010), through Texas General Land Office land surveys in the Matagorda and San Antonio Bays area (White et al. 2002), and anecdotally in the Corpus Christi and Port Aransas areas. Both black mangrove (*Avicennia germinans*) and saltmarsh, composed primarily of smooth cordgrass (*Spartina alterniflora*), are productive estuarine habitats that support a diverse group of benthic organisms. They serve as nursery ground for many species of fish and invertebrates and provide a source of organic nutrients (Chen and Twilley 1999, Stevens et al. 2006). *Avicennia germinans* and *S. alterniflora* are found in similar estuarine locations and can be found in close association (Britton and Morton 1989, Kangas and Lugo 1990). The distribution of these species is largely dependent on climate and salinity (Penfound and Hathaway 1938, Sherrod and McMillan 1985, Kangas and Lugo 1990, Saenger 2002). However, environmental factors such as freshwater inflow, salinity, and nutrient levels can favor the expansion of black mangrove into *Spartina* habitats from which *S. alterniflora* do not often recolonize (Eady 2007).

Abiotic factors play a large role in controlling *Spartina* and *Avicennia* interplay. *Spartina alterniflora* dominate at lower elevations where inundation occurs regularly and soils have more reducing characteristics (Patterson et al. 1993). *Avicennia germinans* dominate in areas of higher elevation with soils that are more oxidized, better drained and have higher salt contents (Patterson et al. 1993). *Spartina alterniflora* is more dependent on freshwater inflow than *Avicennia*. Though both species do well in a wide range of salinity (Chapman 1974, 1976), *Avicennia* has a greater tolerance for increased salinity levels, maintaining production at sa-

linities exceeding 40 (Patterson and Mendelssohn 1991). A severe drought in Louisiana caused significant dieback of *S. alterniflora* stands but did not affect *Avicennia* (McKee et al. 2004). In contrast, *S. alterniflora* is more cold tolerant, rapidly recolonizing after winter freezes cause dieback of *A. germinans* (Stevens et al. 2006).

Avicennia germinans is the only species of mangrove commonly found on the Texas coast, and winter freezes have limited permanent stands of this species to the southern coast of Texas (McMillan 1986, McMillan and Sherrod 1986, Lonard and Judd 1991). However, recent mild winters have allowed the distribution of black mangroves to expand northward along the Gulf of Mexico (GOM) coast (Everitt et al. 2010). The interaction between *A. germinans* and *S. alterniflora* is a complex process. *Avicennia germinans* outshades *S. alterniflora* in times of mild winters but is out competed for space in areas of mixture (Stevens et al. 2006). However, *S. alterniflora* may also facilitate *A. germinans* expansion by trapping *Avicennia* propagules and allowing them to establish.

While much research has been done on black mangrove expansion and the effects of mangrove–saltmarsh competition (Stevens et al. 2006, Everitt et al. 2007, 2008, 2010), little has been done on how the biotic communities compare in Texas. Several studies have investigated how macrofaunal assemblages compare in other parts of the world, though infaunal studies are lacking. Studies done previously have determined that mangroves generally support lower densities of fish and invertebrates than other vegetated habitats (Sheridan and Hays 2003). However, densities of fish and invertebrates in saltmarshes were lower than densities of fish and invertebrates in mangroves along the Australian coast (Bloomfield and Gillanders 2005). Changes in vegetated community composition are expected to have some effects on the benthic communities these biogenic habitats support (Posey 1988). The structurally complex habitat cre-

ated by stem and root structures of A. germinans and S. alterniflora may attract different groups and species of organisms. Mangroves also tend to acidify their substrate, which may also affect community composition (McKee 1993).

This study examined the effects of mangrove expansion on benthic communities. This was accomplished by examining plant composition, benthic community composition, and belowground root biomass. Two sites were examined: Corpus Christi Pass, Corpus Christi, TX where mangroves have become more common in the last decade and the Lower Laguna Madre, South Padre Island, TX where mangroves are well established. This research seeks to provide preliminary results to provide direction for future studies examining the interplay of S. alterniflora and A. germinans habitat on benthic communities. We hypothesize that the more established site of South Padre Island will have greater numbers of species than the newer communities of Corpus Christi, and that A. germinans habitats will have lower diversity of infaunal organisms than S. alterniflora habitats.

Figure 1. A. Map of Texas coastline with experimental sites marked by stars. **B.** Diagram of transect sampling design. Circles represent where cores were taken for community sampling. Plant composition transect were set up in the same way but spaced farther apart. **C.** Comparison of percent cover of Spartina alterniflora and Avicennia germinans along transects at both field sites (r = −0.47). Corpus Christi values are represented by diamonds (n = 9), South Padre Island values are represented by circles (n = 40).

MATERIALS AND METHODS

Wetland Plant Composition

Sites were selected in tidal marsh habitats, one in the Lower Laguna Madre, South Padre Island, TX and one at Corpus Christ Pass in the Upper Laguna Madre, Corpus Christi, TX (Figure 1A). Both were in 10–15 m long sections of mixed habitats of A. germinans and S. alterniflora. Marsh vegetation along the Texas coast exhibits a zonation pattern with S. alterniflora in more submerged areas with A. germinans further inshore from S. alterniflora.

Plant composition was quantified in our field sites. Transects were initiated within the fringing stands of S. alterniflora and extended perpendicular to shore into stands of A. germinans. Three transects were spaced 50 m apart along the shoreline. Percent composition of live wetland plant species was determined every 1.5 m using a 1 m^2 quad-

rat. Data points which contained neither A. germinans nor S. alterniflora coverage were removed from the data set. The data points were then log_{10} transformed to meet assumptions of normality. A correlation between A. germinans and S. alterniflora cover at both the Corpus Christi and South Padre Island sites determined the interaction between species as they compete for space along the coastline.

Belowground Root Biomass

Belowground root biomass was sampled using a core (10 cm diameter x 10 cm depth) at the Corpus Christi site. Logistics prevented belowground root biomass from being examined at the South Padre Island site. Cores were taken in conjunction with plant composition transects but with stations spaced every 25 m. At each of these stations, one core was taken in each habitat: the zone of overlap, A. germinans, and S. alterniflora habitats (n = 3/habitat) and placed in a 250 μm biobag. Cores were about 1 m apart within stations. Samples were washed in the biobag, roots and stems were separated, and then samples were dried in a drying oven

at 50°C for one week. Data were log transformed to meet normality assumptions. Total mass (g), root and stem mass (g), density (#/m²), percent root, and percent stem data were analyzed using a one–way ANOVA with habitat as the fixed effect. Tukey's post–hoc test was used to determine which parameters differed between species.

Benthic Core Sampling

Core samples (10 cm diameter x 10 cm depth) were taken every 3 m along a 15 m section of coast for a total of 6 stations. At each of these stations, one core was taken in the middle of the 3 habitat types (zone of overlap between *A. germinans* and *S. alterniflora, A. germinans*, and *S. alterniflora* habitats) for a total of 6 cores per habitat type (Figure 1B). There was about 1 m between each habitat core within a station and 3 m between station cores. All samples were taken within one week in early fall of 2009. Samples were placed into 250 μm biobags and washed in the field. Samples were fixed in 10% formalin, then washed and transferred to jars of 45% isopropyl alcohol. All organisms were sorted using dissecting microscopes. Polychaetes were identified to family and all other organisms were identified to species.

Species diversity was calculated with the Shannon–Weiner Index and counts were expressed as number/m². Data were log_{10} transformed to meet normality assumptions for parametric statistics. A two–way analysis of variance (ANOVA) with site and habitat as fixed factors was used to analyze species richness, diversity, number of organisms/m², total crustaceans, total mollusks and total polychaetes. If the interaction between site and habitat was not significant, Tukey's post hoc test was used to compare all pairwise differences for both site and habitat (Sokal and Rohlf 1995). All statistical analyses were performed using SAS software version 9.1.3, and $p \leq 0.05$ was considered significant.

Results

Wetland Plant Composition

Avicennia germinans and *S. alterniflora* were inversely correlated with a narrow region of mixing for both sites ($r = -0.47$, $p = 0.001$; Figure 1C). Visual observations at both sites indicate *S. alterniflora* dominate deeper waters along the south Texas coast.

Belowground Root Biomass

Belowground root biomass was not significantly different between habitats for any of the parameters measured (all p > 0.2). This could be because of the small sample size or that belowground structure differs more than the actual biomass.

Benthic Core Sampling

The South Padre Island (SPI) site was significantly higher in organism counts (lognm²), species richness (logS), diversity (logH), and total polychaetes (logtotpoly) than was Corpus Christi Pass (CC) ($F_{1,30} = 5.17$, $p = 0.03$; $F_{1,30} = 12.44$, $p = 0.0014$; $F_{1,30} = 10.81$, $p = 0.0026$; $F_{1,30} = 28.51$, $p < 0.0001$, respectively; Figure 2 A–D). Sites did not differ in total mollusks (logtotmoll) or total crustaceans (logtotcrust) ($F_{2,30} = 2.46$, $p = 0.12$; $F_{2,30} = 0.19$, $p = 0.66$, respectively). There was no significant interaction of site and habitat for any of the parameters measured (lognm²: $F_{2,30} = 1.11$, $p = 0.34$; logH: $F_{2,30} = 0.66$, $p = 0.52$; logS: $F_{2,30} = 0.49$, $p = 0.61$; logtotpoly: $F_{2,30} = 0.36$, $p = 0.70$; logtotmoll: $F_{2,30} = 0.76$, $p = 0.47$; logtotcrust: $F_{2,30} = 0.28$, $p = 0.75$).

Habitat data was combined across sites because there was
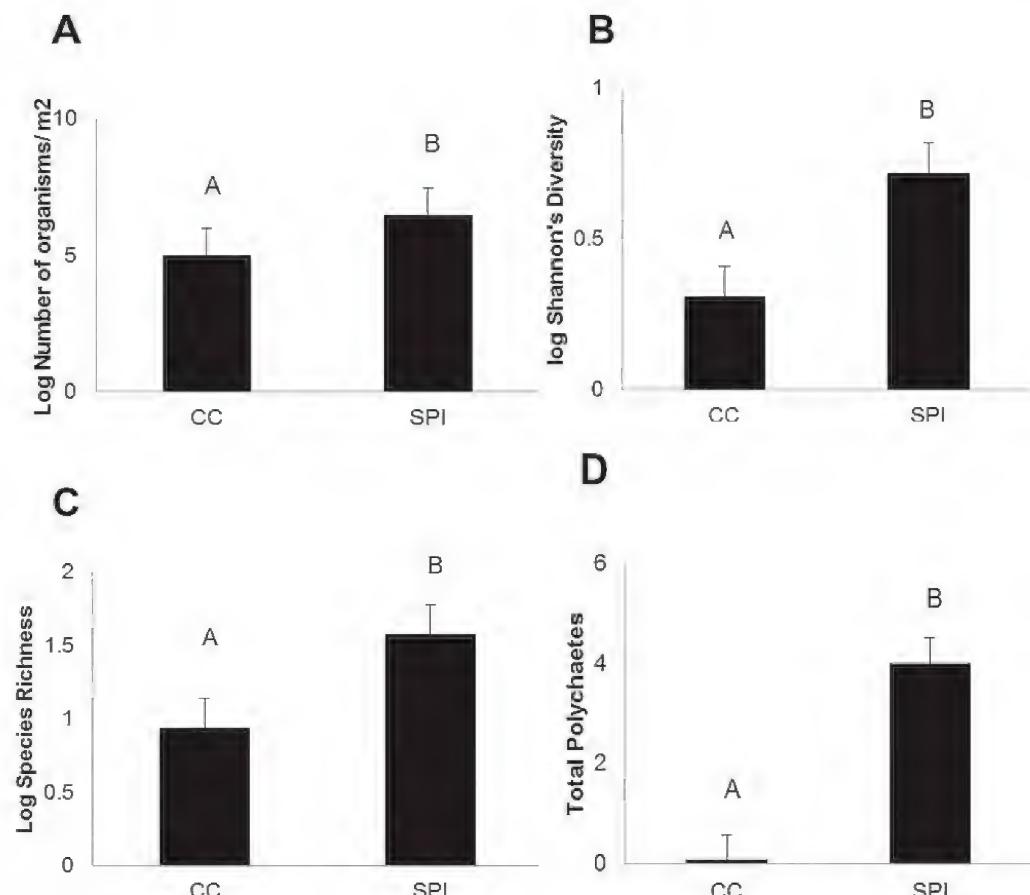
Figure 2. Site differences for benthic community sampling (mean ± sd). Letters represent significant differences among sites (n = 18). **A.** Abundance. **B.** Diversity. **C.** Species richness. **D.** Total polychaetes . CC – Corpus Christi; SPI – South Padre Island.

A

B

C
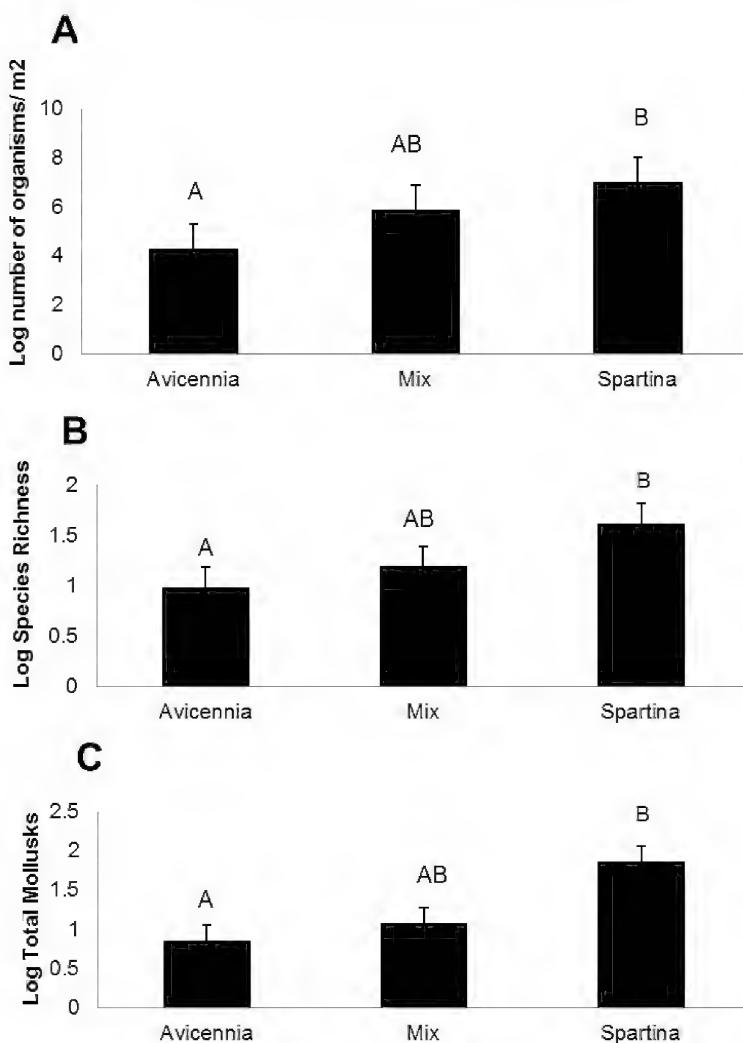
Figure 3. *Habitat differences for benthic community sampling (mean ± sd). Letters represent significant differences among habitats (n = 6); sites were combined for this analysis.* **A.** *Abundance.* **B.** *Species richness.* **C.** *Total mollusks.*

no significant interaction between site and habitat. Habitats differed significantly in organism counts (lognm^2), species richness (logS), and total mollusks (logtotmoll) ($F_{2,30}$ = 5.78, p = 0.0075; $F_{2,30}$ = 3.67, p = 0.03; $F_{2,30}$ = 3.74, p = 0.03, respectively; Figure 3 A–C). *Spartina alterniflora* habitat had significantly higher organism counts, species richness and more mollusks than the *A. germinans* habitat though the mixed habitat did not differ from either of the others. There were no differences between habitats for diversity (logH), total polychaetes (logtotpoly) or total crustaceans (logtotcrust) ($F_{2,30}$ = 2.18, p = 0.13; $F_{2,30}$ = 0.89, p = 0.42; $F_{2,30}$ = 0.07, p = 0.93, respectively).

DISCUSSION

The South Padre Island site was more diverse than the Corpus Christi site. Differences between sites were not unexpected as the South Padre Island site has a longer history of mangrove establishment. The nature of the field sites was also different topographically. For example, the South Padre site had a shallower grade sloping out of the water than the Corpus Christi site which was along a pass cut through Mustang Island. This difference in site topography is because in Corpus Christi the majority of black mangrove and salt marsh overlap occurs along passes or cuts rather than as barriers along the coast. Despite the differences in sites there were consistent differences among habitats.

The zone of habitat overlap was not different in any of the measured parameters from either the *S. alterniflora* or *A. germinans* habitats. This was not surprising because there is very little overlap of the habitats (Figure 1C). *Spartina alterniflora* habitats were more speciose than *Avicennia* habitats, though this may possibly be a function of tidal height. *Spartina* marshes are being pushed into more subtidal habitats as *A. germinans* prefer the higher intertidal areas. Our study sites did not provide a tidal height comparison, as black mangroves were always present higher in the marsh, as is also the case in Louisiana and Florida (Patterson and Mendelssohn 1991). However, a previous study on *S. alterniflora* and *Phragmites auatralis* competition found no changes in *S. alterniflora* benthic communities with tidal height (Posey et al. 2003). It is therefore possible that we are losing infaunal species and diversity in the intertidal zone as *S. alterniflora* is outcompeted. Loss of diversity will lead to less resilient and productive areas that are already subjected to numerous other threats.

Previous studies on the interplay between *A. germinansa* and *S. alterniflora* have shown no differences in ecosystem processes between the ecotones (Perry and Mendelssohn 2009), though there are differences in soil chemistry and fauna (Patterson and Mendelssohn 1991; Stevens et al. 2006). Our data indicates that *A. germinans* and *S. alterniflora* habitats also differ in the benthic infauna they support. Benthic communities provide crucial services to ecosystems (Snelgrove et al., 1997) and changes in the benthic community may affect higher trophic levels, and the community structure of our coastal systems. This study seeks to provide baseline data for future studies conducted on the interplay of *A. germinans* and *S. alterniflora* habitats.

Literature Cited

Britton, J.C. and B. Morton. 1989. Shore Ecology of the Gulf of Mexico. University of Texas Press, Austin, TX, USA, 387 p.

Chapman, V.J. 1974. Salt Marshes and Salt Deserts of the World. Interscience Publishers, New York, NY, USA, 392 p.

Chapman, V.J. 1976. Mangrove Vegetation. J. Cramer Publishers, Vaduz, Liechtenstein, 447 p.

Chen, R. and R.R. Twilley. 1999. A simulation model of organic matter and nutrient accumulation in mangrove wetland soils. Biogeochemistry 44:93–118.

Eady, S. 2007. Effects of snail grazing and environmental factors on the expansion of mangroves into salt marshes. MS thesis. Texas Christian University, Fort Worth, TX, USA, 81p.

Everitt, J.H., C. Yang, K.R. Summy, F.W. Judd, and M.R. Davis. 2007. Evaluation of color–infrared photography and digital imagry to map black mangrove on the Texas gulf coast. Journal of Coastal Research 23:230–235.

Everitt, J.H., C. Yang, S. Sriharan, and F.W. Judd. 2008. Using high resolution satellite imagery to map black mangrove on the Texas gulf coast. Journal of Coastal Research 24:1582–1586.

Everitt, J.H., C. Yang, F.W. Yang, and K.R. Summy. 2010. Use of archive aerial photography for black mangrove populations. Journal of Coastal Research 26:649–653.

Kangas, P.C. and A.E. Lugo. 1990. The distribution of mangrove and saltmarsh in Florida. Tropical Ecology 31:32–39.

Lonard, R.I. and F.W. Judd. 1991. Comparison of the effects of the severe freezes of 1983 and 1989 on native woody plants in the lower Rio Grande Valley, Texas. Southwestern Naturalist 36:213–217.

McKee, K.L. 1993. Soil physico–chemical patterns and mangrove species distribution: Reciprocal effects? Journal of Ecology 81:477–487.

McMillan, C. 1986. Isozyme patterns among populations of black mangrove, *Avicennia germinans,* from the Gulf of Mexico–Caribbean and Pacific Panama. Contributions in Marine Science 29:17–25.

McMillan, C. and C.L. Sherrod. 1986. The chilling tolerance of black mangrove, *Avicennia germinans,* from the Gulf of Mexico coast of Texas, Louisiana and Florida. Contributions in Marine Science 29:9–16.

Patterson, C.S. and I.A. Mendelssohn. 1991. A comparison of physicochemical variables across plant zones in a mangal/salt marsh community in Louisiana. Wetlands 11:139–161.

Penfound, W.T. and E.S. Hathaway. 1938. Plant communities in the marshland of southeastern Louisiana. Ecological Monographs 8:1–56.

Perry, C.L. and I.A. Mendelssohn. 2009. Ecosystem effects of expanding populations of *Avicennia germinans* in a Louisiana salt marsh. Wetlands 29:396–406.

Posey, M.H., T.D. Alphin, D.L. Meyer, and J.M. Johnson. 2003. Benthic communities of common reed *Phragmites australis* and marsh cordgrass *Spartina alterniflora* marshes in Chesapeake Bay. Marine Ecology Progress Series 261: 51–61.

Saenger, P. 2002. Mangrove Ecology, Silviculture, and Conservation. Kluwer Academic Publishers, Dordrecht, The Netherlands, 372 p.

SAS Institute Inc., SAS OnlineDoc 9.1.3, Cary, NC: SAS Institute Inc., 2002–2005.

Sheridan, P. and C. Hays. 2003. Are mangroves nursery habitat for transient fishes and decapods? Wetlands 23:449–458.

Sherrod, C.L. and C. McMillan. 1985. The distribution, history, and ecology of mangrove vegetation along the northern Gulf of Mexico coastal region. Contributions in Marine Science 28:129–140.

Snelgrove, P.V.R., T.H. Blackburn, P.A. Hutchings, D.M. Alongi, J.F. Grassle, H. Hummel, G. King, I. Koike, P.J.D. Lambshead, N.B. Ramsing, V. Solis–Weiss, and D. Freckman. 1997. The importance of marine sediment biodiversity in ecosystem processes. Ambio 26:578–583.

Sokal, R.R. and F.J. Rohlf. 1995. Biometry: The Principles and Practice of Statistics in Biological Research, 3rd ed. W.H. Freeman and Co., New York, NY, USA, 880p.

Stevens, P.W., S.L. Fox, and C.L. Montague. 2006. The interplay between mangroves and saltmarshes at the transition between temperate and subtropical climate in Florida. Wetlands Ecology and Management 14:435–444.

White, W.A., T.A. Tremblay, R.L. Waldinger and T.R. Calnan. 2002. Status and trends of Matagorda Bay and San Antonio Bay. Final Report, contract 01–241–R. Texas General Land Office, Austin, TX, USA, 70 p.

Time-Activity Budgets of Stoplight Parrotfish (Scaridae: *Sparisoma viride*) in Belize: Cleaning Invitation and Diurnal Patterns

Wesley A. Dent
University of Mississippi

Gary R. Gaston
University of Mississippi

SHORT COMMUNICATION

TIME—ACTIVITY BUDGETS OF STOPLIGHT PARROTFISH (SCARIDAE: *SPARISOMA VIRIDE*) IN BELIZE: CLEANING INVITATION AND DIURNAL PATTERNS

Wesley A. Dent* and Gary R. Gaston

*Department of Biology, P.O. Box 1848, University of Mississippi, University, MS 38677 USA; *Corresponding author, email: wesley.a.dent@gmail.com*

INTRODUCTION

Stoplight parrotfish (Scaridae; *Sparisoma viride*) are relatively large Caribbean reef fish, characterized by distinctive coloration and behavior. They are large herbivores (Mumby et al. 2012), and play a major role in coral—reef ecosystems due to their grazing and bioerosion of living reefs (Bruggemann et al. 1996). Like all parrotfish, *S. viride* are protogynous hermaphrodites with 3 distinct life phases: juvenile phase (JP), initial phase adults (IP), and terminal phase males (TP) (van Rooij et al. 1995). While other parrotfish employ a variety of feeding modes including scraping, *S. viride* utilize an excavating foraging strategy (Bruggemann et al. 1994). This behavior leads to bioerosion of the coral reef, and has a significant impact on algal biomass of the ecosystem.

The contribution of parrotfish to coral reef ecosystems is well established (reviewed by Welsh and Bellwood 2012). In the Caribbean, a 1983 disease—induced mass mortality of the sea urchin *Diadema antillarum* resulted in a renewed interest in parrotfish. This die off resulted in more algal—dominated reefs, and left parrotfish as the major grazers of the reef ecosystem (Gardner et al. 2003, Mumby et al. 2006). According to one model, 42% of the reef is grazed every 6 months, which is the level required for an equilibrium state of high coral cover instead of high algae cover. This grazing level was possible with both *Diadema* and parrotfish grazing the reef, but parrotfish alone fall short of the grazing intensity necessary for equilibrium in many reefs (Mumby et al. 2007). Further, *S. viride* populations have declined in recent years (Mumby et al. 2012), yet remain vital members of Caribbean reefs and must be protected as such. They are the largest common parrotfish in Belize. Mumby et al. (2006) analyzed the effect of reserves (preservation areas) on parrotfish—grazing capacity and documented the important role played by *S. viride*. Behavioral studies are valuable for understanding the ecological role of coral reef species such as parrotfish, and aid in conservation and management decisions.

Activity budgets of *S. viride* provide insight into the behavior of the fish, their habitat use, temporal patterns, and time—related energy resources. Reef fish are known to show variation in diel (daytime) activities, especially feeding and cleaning (Zemke—White et al. 2002, Sikkel et al. 2004). Activity budgets have been used to characterize behavior of many species, and are especially helpful in distinguishing daily behavioral differences among age groups (Altmann 1974). A previous activity budget study of *S. viride* in Jamaica (Hanley 1984) demonstrated differences among activities by fish life phase and habitat types. Hanley determined that activities of all 3 phases of *S. viride* were dominated by swimming, hovering, and feeding. Our preliminary studies in Belize corroborated some of the findings by Hanley, but also indicated that *S. viride* spent significant time at cleaning stations, an activity seldom observed in Jamaica. Those preliminary observations led to initiation of this study.

The primary objective of this study was to analyze diurnal behavior of *S. viride* in South Water Caye Marine Reserve (SWCMR), Belize. Time—activity budgets were used to characterize behavior (using swimming, feeding, hovering, sheltering, defecating, and cleaning categories) during the diel cycle, and to illustrate differences among the life phases of this species.

MATERIALS AND METHODS

All 640 observations for this study were collected during May—June 2011 at 8 different shallow reef environments in the South Water Caye Marine Reserve (SWCMR), Belize (16°38' to 16°55'N, 88°02' to 88°13'W). The study area centered around South Water Caye, a 6.1 ha (15—acre) island located about 32 km SE of Dangriga and 1.6 km north of the Smithsonian's Caribbean Coral Reef Ecosystems research facility on Carrie Bow Caye. The SWCMR is located along the Belize Barrier Reef; it supports only a small human population, and marine ecosystems surrounding the island remain relatively undisturbed. The SWCMR was established in 1996, and covers 29,800 ha (United Nations Environment Programme 2011).

The 8 sites of this study (Figure 1) consisted of shallow—reef environments 2.5—4.0 m depth. Most observations
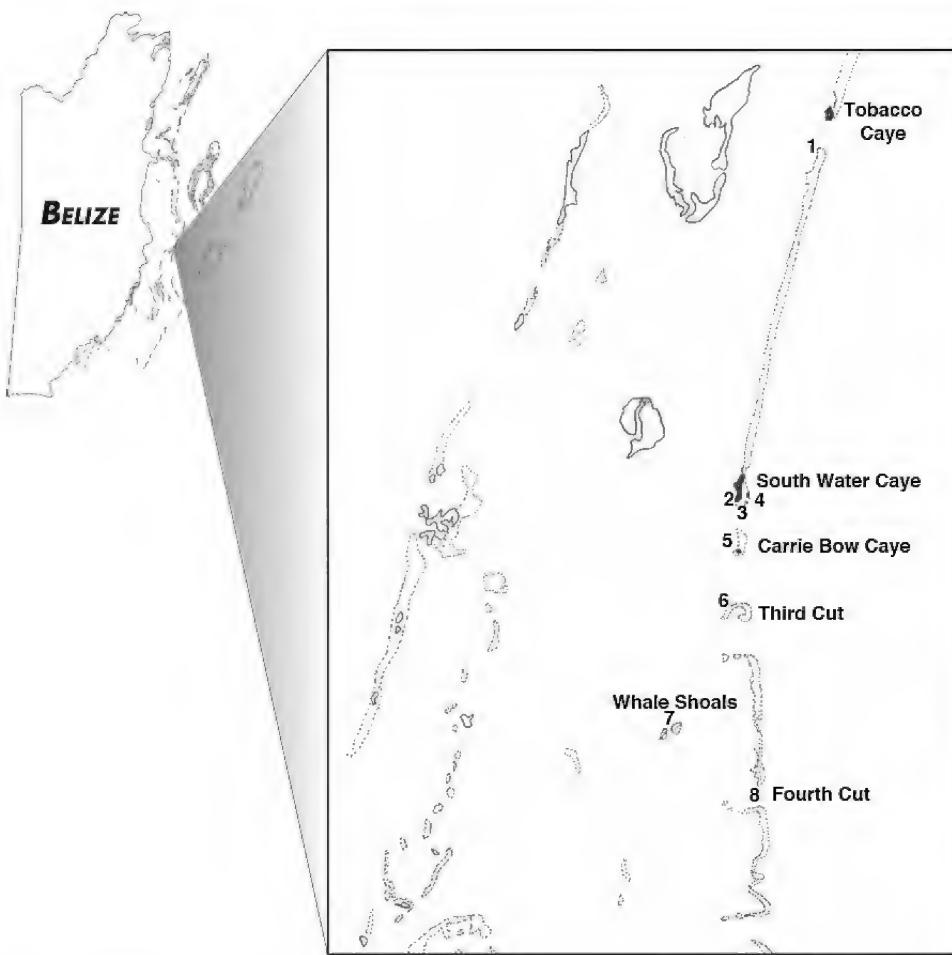
Figure 1: *Study area in the South Water Caye Marine Reserve, Belize (from Gaston et al. 2009).*

were conducted between the hours of 0600–1800 local time. During preliminary analyses that we conducted during 2010 we ascertained that a 20 sec time interval was ideal. Increments of 20 sec was usually the maximum time possible to make observations without disturbance of the fish or the fish moving beyond view, and allowed the opportunity to witness more than one type of behavior. We used instantaneous observations. The fish closest to the observer was selected whenever more than one individual was present. Fish that were deemed disturbed by diver presence were excluded from the data, as were fish that moved beyond view within 20 sec. A Timex waterproof sport watch was used to record time.

The number of seconds per activity in each observation was divided by the total observation time (20 sec) to determine the percentage of time spent on each activity; percentage data were then used for statistical analyses on each behavior (feeding, swimming, sheltering, hovering, defecating, and cleaning) as a separate response variable. For each response variable, we conducted separate multi–factor ANOVAs (using SAS version 9.2) in which the predictor variables were time category, study site, life phase, and water depth. When life phase was a significant predictor, means for pairs of life phase categories were compared by Student's

were made over patch reefs surrounded by sandy bottom off the south end of South Water Caye; only one site included much seagrass (site 2). Although the sites varied in coral makeup and topography, all were characteristic of shallow Caribbean patch reefs. Numerically dominant species of fish included a variety of wrasses, parrotfish, surgeonfish, grunts, snapper, and damselfish. Dominant large coral species were elkhorn coral (*Acropora palmata)*, staghorn coral (*A. cervicornis*), lettuce coral (*Agaricia tenuifolia)*, star corals (especially *Montastrea annularis* and *Stephanocoenia mechelinii*), brain corals (especially *Diploria strigosa*), and a variety of gorgonians (especially *Gorgonia ventalina* and *Briareum asbestinum*).

The behavior of *S. viride* was assessed by classifying their activities into one of 6 categories (Table 1). Activities were recorded by divers who snorkeled at the ocean surface. Data were recorded by pencil on an AquaSketch Minno Wrist Slate. For each observation we recorded: date, start time, location, maximum depth, any pertinent hydrological data, fish developmental stage, and time spent in each behavior. Observation at each site began with the first individual seen, and each individual was observed for 20 sec. Observations

TABLE 1. List of behaviors used for stoplight parrotfish time-activity budget in this study.

Behavior	Description
Swimming (Sw)	Directional locomotion
Feeding (Fe)	Biting, chewing, and scraping food
Hovering (Ho)	No movement or activity other than stabilizing fin movements
Sheltering (Sh)	Time spent inside reef crevices, including while the fish was swimming and stationary
Defecating (De)	Voiding of material
Cleaning (Cl)	Fish oriented itself at an upward angle to invite cleaning from cleaners

T—tests followed by Tukey's Honestly Significant Difference test to determine adjusted p—values. Response variables could not be transformed to achieve normality, so we obtained p—values using randomization tests conducted using a macro wrapper in SAS (modified from Cassel 2002). To determine linear trends in behaviors over the diel cycle, the results from each dive were averaged, and a simple linear regression was conducted for each behavior separately versus time of day (R Statistical Package, version 2.15.1). The data points for each regression are assumed to be independent since different groups of fish likely were observed on each dive. All statistical tests used alpha of 0.05 for significance.

RESULTS

The overall activity budget of *S. viride* was dominated by swimming (37.26%) and feeding (36.91%) (Figure 2A). Hovering, cleaning, sheltering, and defecating were lesser activities (11.90%, 10.51%, 2.23%, 1.19%, respectively).

Feeding increased linearly during the day (Figure 2B) ($p < 0.001$, $r^2 = 0.49$), and was the only activity significantly related to time of day. There was greater cleaning activity during morning (prior to 0800 CST) when morning was used as a category ($p < 0.01$), but the pattern was not linear through the diel period ($p > 0.05$).

The proportion of time spent among the 6 behavior categories differed among the 3 life phases in all behaviors except hovering (Figure 2C). Juvenile phase individuals spent more time swimming than IP ($p < 0.001$) or TP individuals ($p < 0.05$). Initial phase individuals spent more time feeding than TP individuals ($p < 0.01$). Juvenile phase and IP individuals spent less time sheltering than TP individuals ($p < 0.001$ and $p < 0.0001$ respectively). Initial phase and TP individuals spent more time at cleaning stations than JP ($p < 0.01$).

DISCUSSION

The most comprehensive investigation of *S. viride* activities was conducted in Jamaica by Hanley (1984). Most salient in his results, and contrary to ours, was his mention that almost no incidence of cleaning invitation (visits to cleaning stations) was observed. Similarly, *S. viride* in Barbados had very low rates of cleaning invitation (Arnal et al. 2000, 2001), and no other study, even those with early—morning observations, documented cleaning invitation as a common activity of *S. viride*.
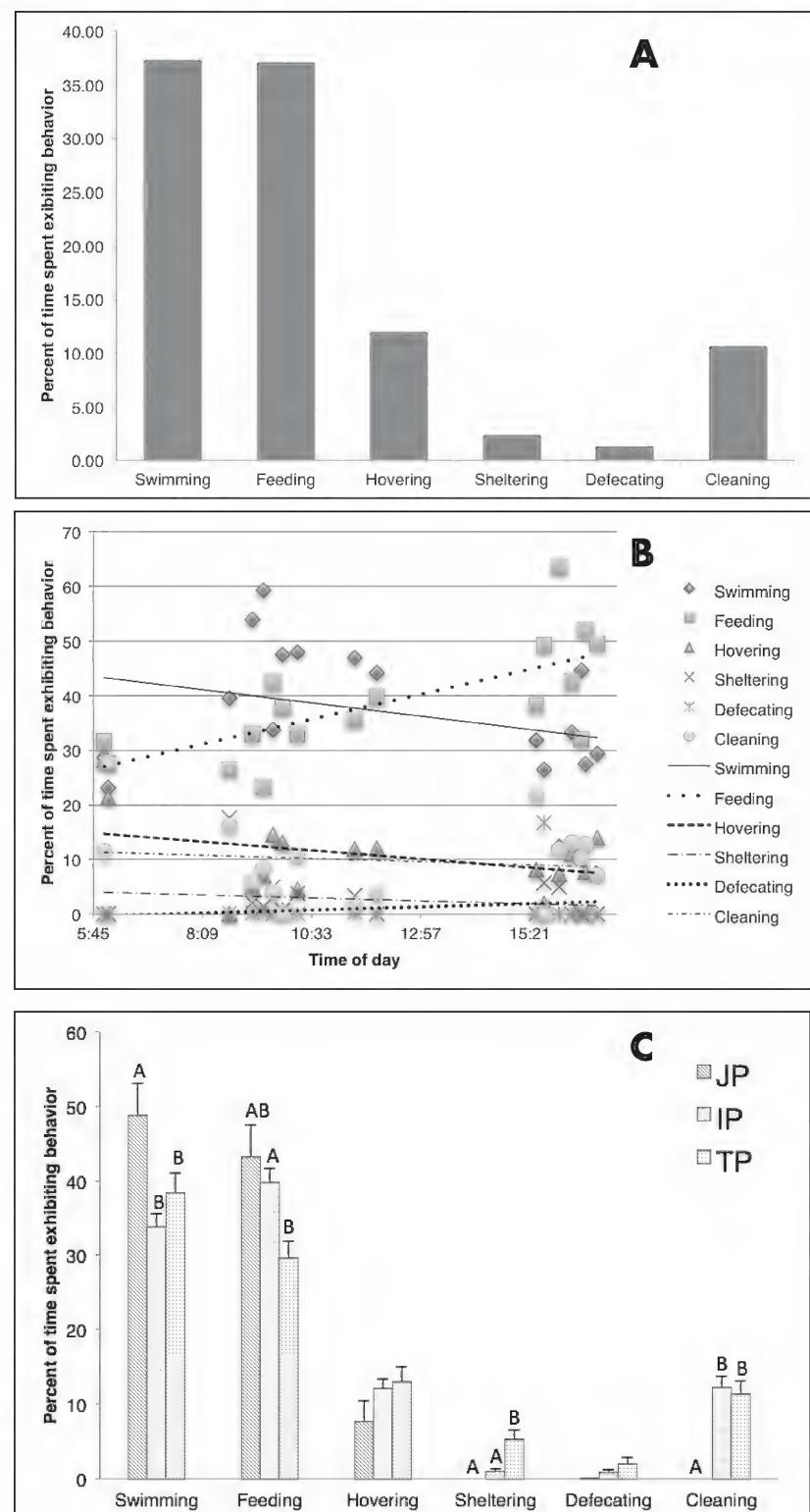

Figure 2. *Time-activity budgets of Sparisoma viride in Belize.* **A.** *Percentage of time spent in behaviors at all observation sites (n = 640).* **B.** *Percentage of time spent in behaviors by time of day. Values for each time are means of observations made during that dive period. Linear regression lines are shown for each activity category. Only the regression line for feeding was significant (percent time = 45.617(time of day) + 15.621, r^2 = 0.49, p < 0.01).* **C.** *Percentage of time (mean ± se) spent in behaviors by life phase. JP - Juvenile Phase; IP - Initial Phase; TP - Terminal Phase. For each response variable, means sharing the same letter were not significantly different from each other. Bars with no letters were not significantly different. Number of observations: n = 76 (JP); n = 357 (IP); n = 207 (TP).*

We found that cleaning invitation was a significant activity of *S. viride* in Belize (10.5% of activities overall). Cleaning invitation was highest during morning, but did not decrease linearly thereafter. Rather, cleaning activity decreased during midday, but trended higher later. Thus, *S. viride* spent significantly more time at cleaning invitation during mornings (before 0800; 20.4%) and less often thereafter (8.1%). These patterns were similar to activities reported for yellowtail damselfish in Barbados, the first study to estimate total diel cleaning time budgets in a marine reef fish (Sikkel et al. 2005). Sikkel et al. (2005) also reported about 20% of early–morning time spent with cleaners.

We considered 3 hypotheses to explain the differences observed related to cleaning behavior of *S. viride* between our study and those of Hanley (1984) and Arnal et al. (2000, 2001). First, we considered that methods between the 2 studies may have accounted for the differing results, but it seems unlikely that methods could yield such different results in fish behavior. We also considered whether variations in mucous load caused *S. viride* to visit cleaning stations more often, but there is no evidence that mucous loads of a species would vary across the Caribbean, nor would non–cleaning activities vary due to mucous loads. Finally, we investigated if habitat could account for higher levels of cleaning invitation. Even though many aspects of physical habitat were similar between Belize and Jamaica (both were shallow reefs), parasite loads are known to change both spatially and temporally by habitat (Grutter 1994, Sikkel et al. 2009). Previous studies reported that *S. viride* carry at least moderate ectoparasite loads (Soares et al. 2007) and visit cleaning stations for parasite removal (Grutter 2001). Thus, we submit that differing parasite burdens among the habitats led to higher cleaning activity, despite having no direct data on parasite loads for the regions.

Sparisoma viride activity budgets are not known for many regions of the Caribbean, but our observations and those in Bonaire (Bruggemann et al. 1994) established that feeding begins about an hour after sunrise. The feeding activity of herbivorous fish is typically highest in the late morning and afternoon and lower in the early morning (Zemke–White et al. 2002). Our data indicated that *S. viride* follow this trend as well. Our earliest records were at 0600 (~ 45 min after sunrise) and feeding was seen to increase linearly throughout the day (Figure 2B).

Hovering may be related to territorial behavior, especially in TP parrotfish (Bruggemann et al. 1994). *Sparisoma viride*

hovering close to the bottom may have been exhibiting a horizontal cleaning invitation to cleaner shrimp (Huebner and Chadwick 2012). However, none of the fish that we observed were being cleaned by shrimp, and cleaning shrimp were not observed on those reefs during invertebrate surveys (G. Gaston, personal observation). Terminal phase fish in our study spent more time hovering than did other life phases.

Our study corroborated Hanley's (1984) report that swimming, feeding, and hovering were the dominant activities throughout the day, even though the percentages of each activity varied between the studies. Hanley (1984) distinguished between individuals by size. Small individuals spent most time hovering (40–50%), swimming (20–30%), and feeding (17–23%); large individuals by swimming (40–52%), hovering (21–30%), and feeding (9–14%). *Sparisoma viride* in our study spent more time feeding and less time hovering than Hanley (1984) observed. We found that juvenile individuals spent more time in active behaviors (swimming and feeding), and less time in other behaviors (hovering, sheltering, and cleaning) (Figure 2C). This makes sense physiologically, because smaller individuals generally have higher metabolic demands than larger fish, but Bruggemann et al. (1994) reported that juvenile *S. viride* in Bonaire spent less time swimming between food patches and more time hovering, presumably looking for animal prey. Bruggemann et al. (1994) conducted comprehensive analyses of food intake and assimilation efficiencies of *S. viride*, which accounted for the discrepancy. They found that daily number of bites decreased with fish size, and number of bites also depended on life phase and foraging depth. Juvenile *S. viride* must grow quickly in habitats with high predation pressure (van Rooij et al. 1995), and they attain that rapid growth by higher assimilation efficiencies than adults (Bruggemann et al. 1994). Their success at growth apparently is unrelated to cleaning symbiosis. We seldom observed juvenile *S. viride* being cleaned, and previous research concluded that juvenile fish are rarely cleaned (Côté et al. 1998).

In summary, we used time activity budgets to discern significant patterns in activities of *S. viride* in Belize related to time of day, life phase, and behavior. We propose that early morning visits to cleaning stations likely were related to ectoparasite burden and/or mucous load. More conclusive evidence awaits future investigations of *S. viride*'s cleaning symbiosis, parasite loads, and diurnal activities.

Literature Cited

Altmann, J. 1974. Observational study of behavior: sampling methods. Behaviour 49:227–267.

Arnal, C., I.M. Côté, P. Sasal, and S. Morand. 2000. Cleaner–client interactions on a Caribbean reef: influence of correlates of parasitism. Behavioral Ecology and Sociobiology 47:353-358.

Arnal, C., I.M. Côté, and S. Morand. 2001. Why clean and be cleaned? The importance of client ectoparasites and mucus in a marine cleaning symbiosis. Behavioral Ecology and Sociobiology 51:1-7.

Bruggemann, J.H., M.J.H. van Oppen, and A.M. Breeman. 1994. Foraging by the stoplight parrotfish, *Sparisoma viride*. I. Food selection in different, socially determined habitats. Marine Ecology Progress Series 106:41–55.

Bruggemann, J.H., A.M. van Kessel, J.M. van Rooij, and A.M. Breeman. 1996. Bioerosion and sediment ingestion by the Caribbean parrotfish *Scarus vetula* and *Sparisoma viride*: implications of fish size, feeding mode and habitat use. Marine Ecology Progress Series 134:59–71.

Cassel, D.L. 2002. A randomization test wrapper for SAS PROCs. SAS Users' Group International Proceedings 27:251. http://www2.sas.com/proceedings/sugi27/p251–27.pdf (viewed on 1/17/2013).

Côté, I.M., C. Arnal, and J. D. Reynolds. 1998. Variation in posing behaviour among fish species visiting cleaning stations. Journal of Fish Biology 53:256–266.

Gardner, T.A., I.M. Côté, J.A. Gill, A. Grant, and A.R. Watkinson. 2003. Long–term region–wide declines in Caribbean corals. Science 301:958–960.

Gaston, G.R., C. Eason, G. Eason, J. Janaski, and M.A. Ballas. 2009. Seagrass loss in Belize: Studies of turtlegrass (*Thalassia testudinum*) habitat using remote sensing and ground–truth data. Gulf and Caribbean Research 21:1–8.

Grutter, A.S. 1994. Spatial and temporal variations of the ectoparasites of seven reef fish species from Lizard Island and Heron Island, Australia. Marine Ecology Progress Series 115:21–30.

Grutter, A.S. 2001. Parasite infection rather than tactile stimulation is the proximate cause of cleaning behaviour in reef fish. Proceedings of the Royal Society B: Biological Sciences 268:1361–1365.

Hanley, F. 1984. Time–budgeting and foraging strategy of the stoplight parrotfish *Sparisoma viride* Bonnaterre, in Jamaica. Journal of Experimental Marine Biology and Ecology 83:159–177.

Mumby, P.J., C.P. Dahlgren, A.R. Harborne, C.V. Kappel, F. Micheli, D.R. Brumbaugh, K.E. Holmes, J.M. Mendes, K. Broad, J.N. Sanchirico, K. Buch, S. Box, R.W. Stoffle, and A.B. Gill. 2006. Fishing, trophic cascades, and the process of grazing on coral reefs. Science 311:98–101.

Mumby, P.J., A. Hastings, and H.J. Edwards. 2007. Thresholds and the resilience of Caribbean coral reefs. Nature 450:98–101.

Mumby, P.J., R.S. Steneck, A.J. Edwards, R. Ferrari, R. Coleman, A.R. Harborne, and J.P. Gibson. 2012. Fishing down a Caribbean food web relaxes trophic cascades. Marine Ecology Progress Series 445:13-24.

Sikkel, P.C., K.L. Cheney, and I.M. Côté. 2004. In situ evidence for ectoparasites as a proximate cause of cleaning interactions in reef fish. Animal Behavior 68:241–247

Sikkel, P.C., S.E. Herzlieb, and D.L. Kramer. 2005. Compensatory cleaner–seeking behavior following spawning in female yellowtail damselfish. Marine Ecology Progress Series 296:1–11.

Sikkel, P.C., D. Nemeth, A. McCammon, and E.H. Williams, Jr. 2009. Habitat and species differences in prevalence and intensity of *Neobenedenia melleni* (Monogenea: Capsalidae) on sympatric Caribbean surgeonfishes (Acanthuridae). Journal of Parasitology 95:63–68

Soares, M.C., S.C. Cardoso, and I.M. Côté. 2007. Client preferences by Caribbean cleaning gobies: food, safety or something else? Behavioral Ecology and Sociobiology 61:1015-1022.

United Nations Environment Programme. 2011. Belize Barrier Reef Reserve System, Belize. http://www.eoearth.org/article/Belize_Barrier_Reef_Reserve_System,_Belize (viewed on 03/28/2012).

van Rooij, J.M., J.H. Bruggemann, J.J. Videler, and A.M. Breeman. 1995. Plastic growth of the herbivorous reef fish *Sparisoma viride*: field evidence for a trade–off between growth and reproduction. Marine Ecology Progress Series 122:93–105.

Welsh, J.Q. and D.R. Bellwood. 2012. How far do schools of roving herbivores rove? A case study using *Scarus rivulatus*. Coral Reefs 31:991–1003.

Zemke–White, L.W., J.H. Choat, and K.D. Clements. 2002. A re–evaluation of the diel feeding hypothesis for marine herbivorous fishes. Marine Biology 141:571–579.

First Record of a Nurse Shark, *Ginglymostoma cirratum*, within the Mississippi Sound

Jill M. Hendon
University of Southern Mississippi

Eric R. Hoffmayer
National Marine Fisheries Service

William B. Driggers III
National Marine Fisheries Service

SHORT COMMUNICATION

FIRST RECORD OF A NURSE SHARK, *GINGLYMOSTOMA CIRRATUM*, WITHIN THE MISSISSIPPI SOUND

Jill M. Hendon[1], Eric R. Hoffmayer[2], and William B. Driggers III[2]

[1]Center for Fisheries Research and Development, Gulf Coast Research Laboratory, The University of Southern Mississippi, 703 East Beach Drive, Ocean Springs, MS 39564 USA; [2]National Marine Fisheries Service, Southeast Fisheries Science Center, Mississippi Laboratories, P.O. Drawer 1207, Pascagoula, MS 39567 USA; [1]Corresponding author e–mail: jill.hendon@usm.edu

INTRODUCTION

Among habitats within the northern Gulf of Mexico (GOM), the Mississippi Sound (hereafter Sound) is considered one of the most dynamic and biologically productive due to tidal forces and riverine inputs (Gunter 1963, Kjerfve 1986). As such, a diverse assemblage of organisms is known to inhabit the region, including 13 shark species (Christmas et al. 1973, Parsons and Hoffmayer 2007). The predominant species, Atlantic sharpnose (*Rhizoprionodon terraenovae*), blacktip (*Carcharhinus limbatus*) and finetooth (*C. isodon*) sharks, are known to occur in waters with widely ranging abiotic characteristics (Parsons and Hoffmayer 2007). Other species, including the blacknose (*C. acronotus*), tiger (*Galeocerdo cuvier*) and great hammerhead (*Sphyrna mokarran*) sharks, are associated with times of reduced riverine input and/or increased intrusion of marine water into the Sound, often associated with an incoming tide (J. Hendon, unpublished data).

Of all the coastal shark species in the northern GOM, the nurse shark, *Ginglymostoma cirratum*, is one of the few species not documented in the Sound. Unlike most coastal sharks, nurse shark distribution is confined to a relatively specific suite of abiotic characteristics. Hannan et al. (2012) demonstrated that the distribution of nurse sharks in the northern GOM is primarily limited to hard bottom habitats with relatively high temperature, salinity, dissolved oxygen, and water clarity. Herein, we provide support for the findings of Hannan et al. (2012) and report the first documented occurrence of a nurse shark in the Sound.

MATERIALS AND METHODS

Gillnet, handline, and bottom longline surveys were conducted monthly from March through October in the coastal waters of Alabama, Mississippi, and Louisiana from 1998–2012, 2004–2012 and 2007–2012, respectively (for gear configuration and deployment see: Parsons and Hoffmayer 2007 for gillnet, Ulrich et al. 2007 for handline, and Drig-

gers et al. 2008 for longline). A bottom water sample was collected at each station, using a Van Dorn water sampler (Wildlife Supply Company) and abiotic conditions (temperature (°C), salinity, and dissolved oxygen (mg/L)) were measured using a YSI–85 (Yellow Springs Instruments Inc.) meter. Additionally, water clarity (m) was determined using a Secchi disk. Upon capture, the species and sex of each shark was determined prior to obtaining a precaudal (PCL) and stretch total length (STL) measurement to the nearest mm. Sharks were externally tagged and released at the capture site.

RESULTS AND DISCUSSION

Over the course of the survey, 345 gillnet, 396 handline, and 349 bottom longline stations were sampled within the Sound and adjacent waters (Figure 1). During these surveys, 54,700 hooks were set and 63,135 m of gillnet were deployed, resulting in the capture of 12,447 sharks. Of the sharks captured, all were species of the families Carcharhinidae and Sphyrnidae, with one exception. On 9 July 2009, a female nurse shark was caught by bottom longline in the Sound, northwest of Horn Island, Mississippi at 30°16.08'N, 88°49.20'W (Figure 1). The shark was in excellent condition and measured 1555 mm PCL and 2227 mm STL. Based on the minimum size at maturity of 2220 mm STL for female nurse sharks reported by Castro (2000), this specimen was mature or in late stages of puberty; however, no internal examination was conducted. After length measurements were obtained, the shark was externally tagged and released at the location of capture.

Water temperature, salinity, dissolved oxygen, and clarity at the capture location was 24.2°C, 36.2, 6.4 mg/L, and 3.4 m, respectively. These values were similar to what Hannan et al. (2012) found characteristic of nurse shark habitat; however, the salinity and water clarity values were atypical for the Sound and were the highest values recorded for this
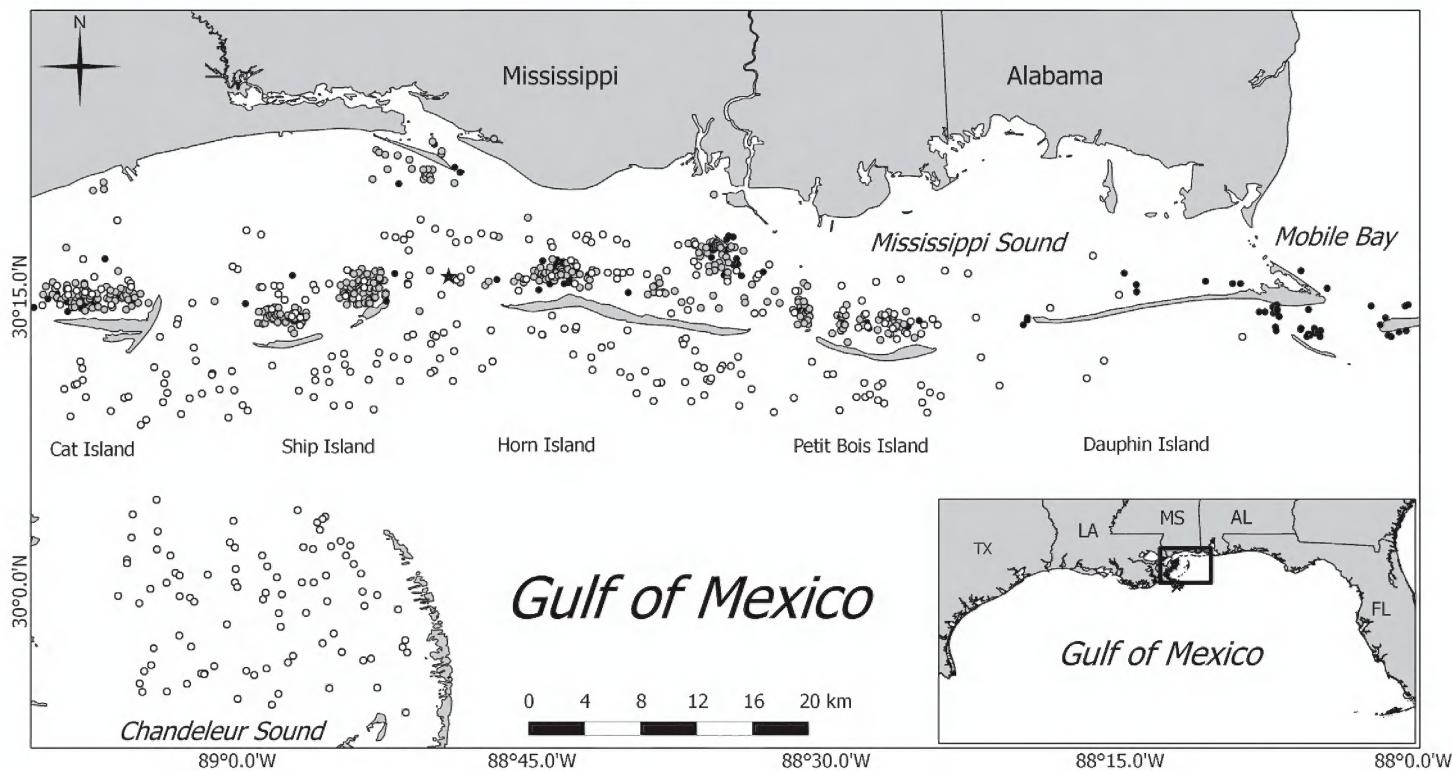
Figure 1. Map of the shark sampling sites from 1998-2012. Gillnet (black circles, 345 sets from 1998-2012), handline (gray circles, 396 sets from 2004-2012), and bottom longline (white circles, 349 sets from 2007-2012) gear was used to capture sharks in the Mississippi Sound and adjacent waters. Handline and gillnet sets were often conducted at the same station resulting in the overlap of many symbols. Black star denotes the location where the nurse shark, Ginglymostoma cirratum, was captured by bottom longline.

region throughout all survey years. During the 48 other gear deployments conducted in close proximity to the nurse shark catch site, the mean (± se) bottom salinity was 27.2 ± 0.5 (range: 16.9 – 32.9) and water clarity was 1.4 ± 0.1 m (range: 0.07 – 2.4 m). As the capture location was just north of the pass between Horn and East Ship Islands, the atypical abiotic conditions were most likely related to an offshore water mass entering the Sound.

Hannan et al. (2012) reported that the majority of nurse sharks in the northern GOM occur east of Mobile Bay, Alabama (~88° west longitude) and suggested that the low occurrence to the west results from the relatively limited amount of hard bottom habitat such as gravel, rock and sand. In the Sound, mud is the dominant sediment type; however, water entering and leaving the Sound must move between the barrier islands which accentuates tidal velocities and leaves sand as the predominant substrate in these passes (Otvos 1973). The atypical water mass and the patch of sand bottom most likely attracted the nurse shark to enter the Sound. Thus, it is possible that nurse sharks occur more frequently in association with other patchy hard bottom habitats off the coasts of Alabama, Louisiana and Mississippi than is currently known.

Literature Cited

Castro, J.I. 2000. The biology of the nurse shark, *Ginglymostoma cirratum*, off the Florida east coast and the Bahama Islands. Environmental Biology of Fishes 58:1–22.

Christmas, J.Y., L.M. Eleuterius, W.W. Langley, H.M. Perry, and R.S. Waller. 1973. Phase IV: Biology. In: J.Y. Christmas, ed. Cooperative Gulf of Mexico Estuarine Inventory and Study, Mississippi. Mississippi Marine Conservation Commission, Gulf Coast Research Laboratory, Ocean Springs, MS, USA, p. 141–434.

Driggers, W.B. III, G.W. Ingram Jr., M.A. Grace, C.T. Gledhill, T.A. Henwood, C.N. Horton, and C.M. Jones. 2008. Pupping areas and mortality rates of young tiger sharks *Galeocerdo cuvier* in the western North Atlantic Ocean. Aquatic Biology 2:161–170.

Gunter, G. 1963. The fertile fisheries crescent. Journal of the Mississippi Academy of Sciences 9:286–290.

Hannan, K.M., W.B. Driggers III, D.S. Hankiso, L.M. Jones, and A.B. Canning. 2012. Distribution of the nurse shark, *Ginglymostoma cirratum*, in the northern Gulf of Mexico. Bulletin of Marine Science 88:73–80.

Kjerfve, B. 1986. Comparative oceanography of coastal lagoons. In: D.A. Wolfe, ed. Estuarine Variability. Academic Press, New York, NY, USA, p. 63–81.

Otvos, E. G. 1973. Phase III: Sedimentology. In: J.Y. Christmas, ed. Cooperative Gulf of Mexico Estuarine Inventory and Study, Mississippi. Mississippi Marine Conservation Commission, Gulf Coast Research Laboratory, Ocean Springs, MS, USA, p. 124–137.

Parsons, G.R. and E.R. Hoffmayer. 2007. Identification and characterization of shark nursery grounds along the Mississippi and Alabama Gulf Coasts. In: C.T. McCandless, N.E. Kohler, and H.L. Pratt Jr., eds. Shark Nursery Grounds of the Gulf of Mexico and the East Coast Waters of the United States. American Fisheries Society Symposium 50. American Fisheries Society, Bethesda, MD, USA, p. 301–316.

Ulrich, G.F., C.M. Jones, W.B. Driggers III, J.M. Drymon, D. Oakley, and C. Riley. 2007. Habitat utilization, relative abundance, and seasonality of sharks in the estuarine and nearshore waters of South Carolina. In: C.T. McCandless, N.E. Kohler, and H.L. Pratt Jr., eds. Shark Nursery Grounds of the Gulf of Mexico and the East Coast Waters of the United States. American Fisheries Society Symposium 50. American Fisheries Society, Bethesda, MD, USA, p. 125–139.